Tennessee Valley Authority
Office of Natural Resources
Division of Air and Water Resources
Water Systems Development Branch

BROWNS FERRY NUCLEAR PLANT

LOW-LEVEL RADWASTE STORAGE FACILITY

GROUND-WATER PATHWAY ANALYSIS

Report No. WR28-1-67-116

Prepared by
J. Mark Boggs

Norris, Tennessee
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EXECUTIVE SUMMARY

The proposed low-level radwaste storage facility (LLRWSF) at Browns Ferry Nuclear Plant is underlain by soils having low hydraulic conductivity and high sorptive capacity which greatly reduce the risks associated with a potential contaminant excursion. A conservative ground-water pathway accident analysis using flow and solute transport modeling techniques indicates that without interdiction the concentrations of the five radionuclides of concern (Sr-90, Cs-137, Cs-134, Co-60, and Mn-54) would be well below 10 CFR Part 20 criteria at downgradient receptors. These receptors include a possible future private water well located near the eastern site boundary and Wheeler Reservoir.

Because of uncertainties associated with some of the site geo-hydrologic data a parametric analysis was performed to bracket the range of possible impacts. In addition to demonstrating that the impacts under expected and worst-case accident conditions would be insignificant, the parametric analysis indicates the results obtained using the predictive model to be credible despite input data uncertainties. These studies show that the overall rate of contaminant transport is governed by factors which control the rate of movement through the soil zone. This implies that in planning data acquisition programs for future contaminant transport analyses in similar geohydrologic environments, measurements of soil properties and net ground-water recharge rates should be emphasized.

Routine ground-water monitoring is not recommended at the LLRWSF except in the unlikely event of an accident. While the analysis indicates that the existing monitor well network designed for the originally planned long-term LLRWSF is not the most effective monitor system for the

current temporary emergency storage facility, no additional monitor wells are recommended at this time. The extremely slow rate of contaminant movement would allow sufficient time to install additional wells (if necessary) should an accident occur. Accident mitigation is unwarranted from a technical viewpoint. However, if mitigation is necessary for other reasons, removal of contaminated soils is judged the most cost-effective method.

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INTRODUCTION

In 1980 the decision was made within TVA to store low-level radioactive waste generated by operations at Browns Ferry Nuclear Plant onsite in above-ground reinforced concrete storage modules. Twenty-five modules were originally planned for an area located near the eastern plant site boundary (Figure 1). Wastes were to be stored onsite for the life of the plant (approximately 40 years) with the decision as to final disposition of wastes deferred until some future date.

Although the waste disposal facility was designed for maximum assurance of waste containment, the politically sensitive nature of radioactive waste disposal dictated that a state-of-the-art assessment of potential impacts which might result from accidents be performed to allay any doubts concerning the safety of the facility. Part of the overall safety assessment included a ground-water pathway impact analysis of an undetected contaminant release through a crack in the floor of a storage module. This report documents that analysis which was initiated in June 1981.

The purpose of the study is to determine the potential impacts associated with an accidental release of liquid contaminants from the storage facility. Ground-water flow and solute transport modeling techniques are used to predict the migration and fate of five radiological contaminants associated with the stored wastes. The specific objectives are to estimate the maximum radionuclide concentrations at two possible receptors of contaminated ground water, i.e., Wheeler Reservoir and a possible future water supply well located down-gradient from the LLRWSF near the eastern plant site boundary. In addition the temporal and spatial distributions of contaminants derived from the predictive models are to be utilized in defining future monitoring and accident mitigation needs.

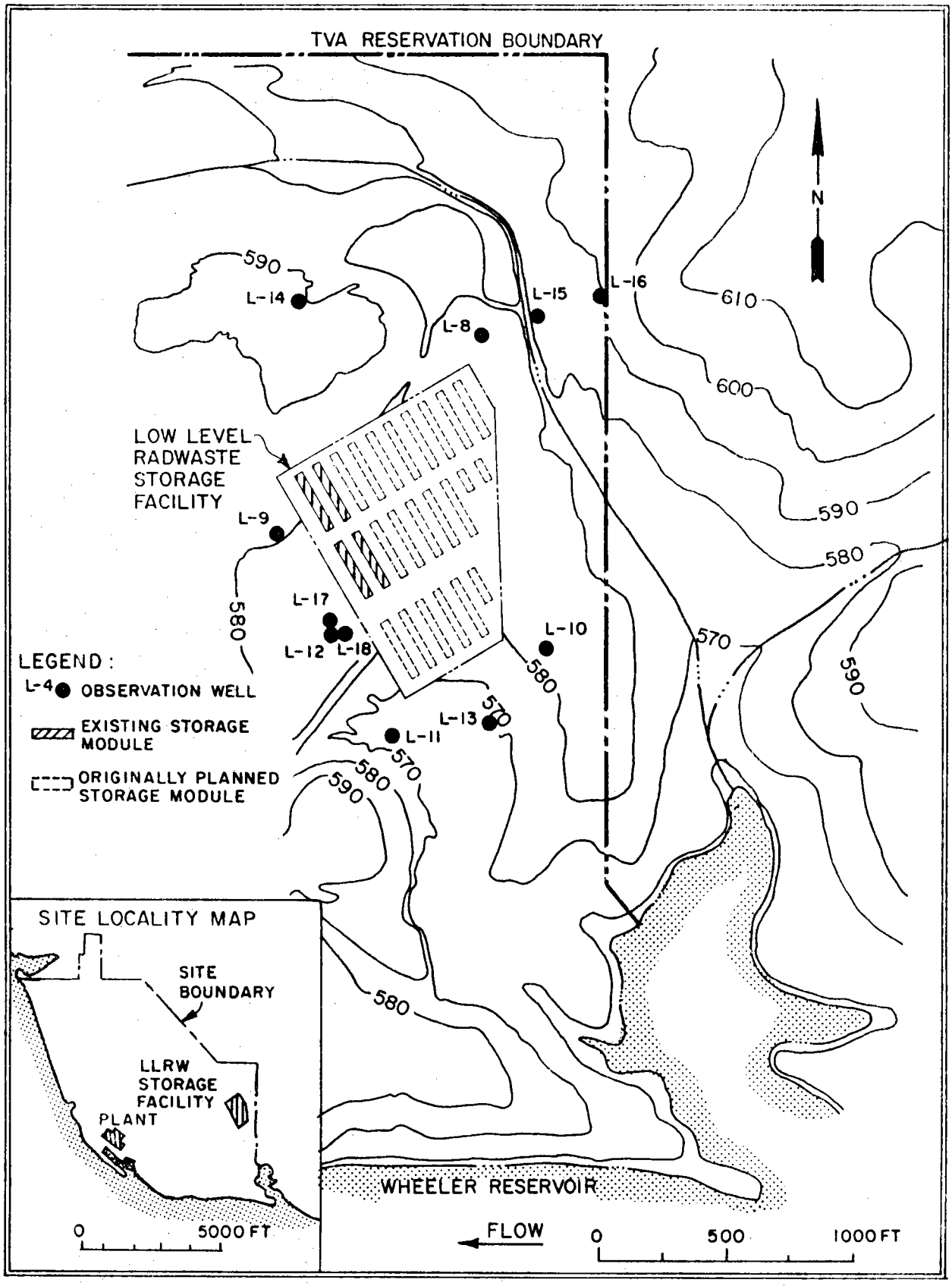


FIGURE 1: Site Location Map

During the course of the analysis, TVA plans concerning low-level radwaste disposal were revised. In January 1981 the number of storage modules was reduced to 14. The revised plan called for radwaste to be temporarily stored for periods of up to five years in modules after which time it would be removed for permanent disposal elsewhere. Most recently TVA announced in June 1982 intentions to dispose of the low-level radwastes generated by all its nuclear plants at regional facilities rather than at onsite facilities. Wastes will be stored onsite only on a temporary emergency basis in the four existing modules pending transport to offsite disposal facilities.

While this decision eliminates much of the impetus for the study, the analysis warrants documentation for several reasons. First, since the storage modules are likely to contain wastes for short periods of time, the results of the study still serve to demonstrate adequate protection to local ground water and surface water users in the event of an accidental release. Second, as the first study of its kind in TVA, it illustrates how mathematical ground-water models can be used to assess the impacts of nonradiological as well as radiological waste disposal. And third, certain general conclusions may be derived from the study concerning modeling and data needs for future assessments of surface waste disposal under similar geohydrologic conditions.

SITE GEOHYDROLOGIC CONDITIONS

The LLRWSF site is underlain by residual and alluvial soils consisting predominantly of clay with minor amounts of silt, sand, and chert gravel. Total soil thickness ranges from 37 to 50 feet and averages approxi-

mately 40 feet. The hydraulic conductivity of the soil was determined through laboratory tests of 15 undisturbed core samples in accordance with procedures outlined in TVA (1978). The saturated hydraulic conductivity of soil averages 3.7E-08 ft/s (1.1E-06 cm/s) which is typical of clay soil. A statistical summary of the physical and hydraulic properties of soils tested as part of site foundation and geohydrologic investigations is presented below in Table 1.

TABLE 1: Soil Properties

Parameter	Minimum	Mean	Maximum	Standard Deviation	No. of Samples
Porosity	0.40	0.46	0.55	0.04	97
Density (lb/ft^3)	76.7	91.6	107.2	4.5	97
Saturated Hydraulic Conductivity (ft/s)	8.2E-10	3.7E-08	2.3E-07	2.0E-07	15
Natural Saturation (%)	75	93	100	5	97

Note: Soil properties compiled from TVA (1980a) and supplemental data provided in memorandum from R. O. Lane to E. E. Driver dated July 2, 1981.

Soil deposits are underlain by the Fort Payne and Tuscumbia formations (Mississippian age) consisting of cherty limestone and dolomite. Thickness of these materials is believed to be in excess of 200 feet. Top of rock is relatively uniform ranging from El. 532 to 542 ft over the facility foundation investigation area (TVA, 1980a). Exploratory boreholes at the site indicate presence of solution cavities within bedrock at several locations. However, seismic refraction profiles across the area indicated no large near-surface cavities or buried sinkholes (TVA, 1980b).

The Fort Payne and Tuscumbia formations comprise an aquifer of major regional importance. The hydraulic properties of this unit have not been measured at the LLRWSF site; however, regionally the transmissivity ranges from 1000 to 3000 ft^2/d, and averages approximately 2000 ft^2/d (Moran, 1977). The porosity of the Fort Payne/Tuscumbia aquifer is estimated to be approximately 0.10. The aquifer is underlain by a sequence of shales, sandstones, and limestones. These units are not germane to the present study because they occur so deep, and thus are not considered further.

The direction of ground-water movement at the nuclear plant site is generally toward the Tennessee River. However, local ground-water movement in the vicinity of the LLRWSF is considerably more complex. Figure 2 shows the potentiometric surface configuration in April 1980. The LLRWSF is situated over a localized mound in the water table. This mounded condition is due to the relatively high topographic position of the area and to complex subsurface drainage conditions. As shown in Figure 1, the site is bounded to the north and east by an ephemeral drainage channel which discharges into a small embayment off the Tennessee River southeast of the site. This feature appears to mark the approximate alignment of subsurface drainage channels in the underlying carbonate rock. This condition is inferred from the presence of surface depressions (sinkholes) in preconstruction topography at the northern end of the drainage channel and from the relatively low hydraulic gradients along this linement. (Low gradients can indicate a zone of relatively high permeability resulting from solution cavity development.) Solution cavity development along this trend is also indicated in the observation well hydrographs shown in Figure 3. Hydrographs for wells L8, L15, and L16 located on the drainage trend all

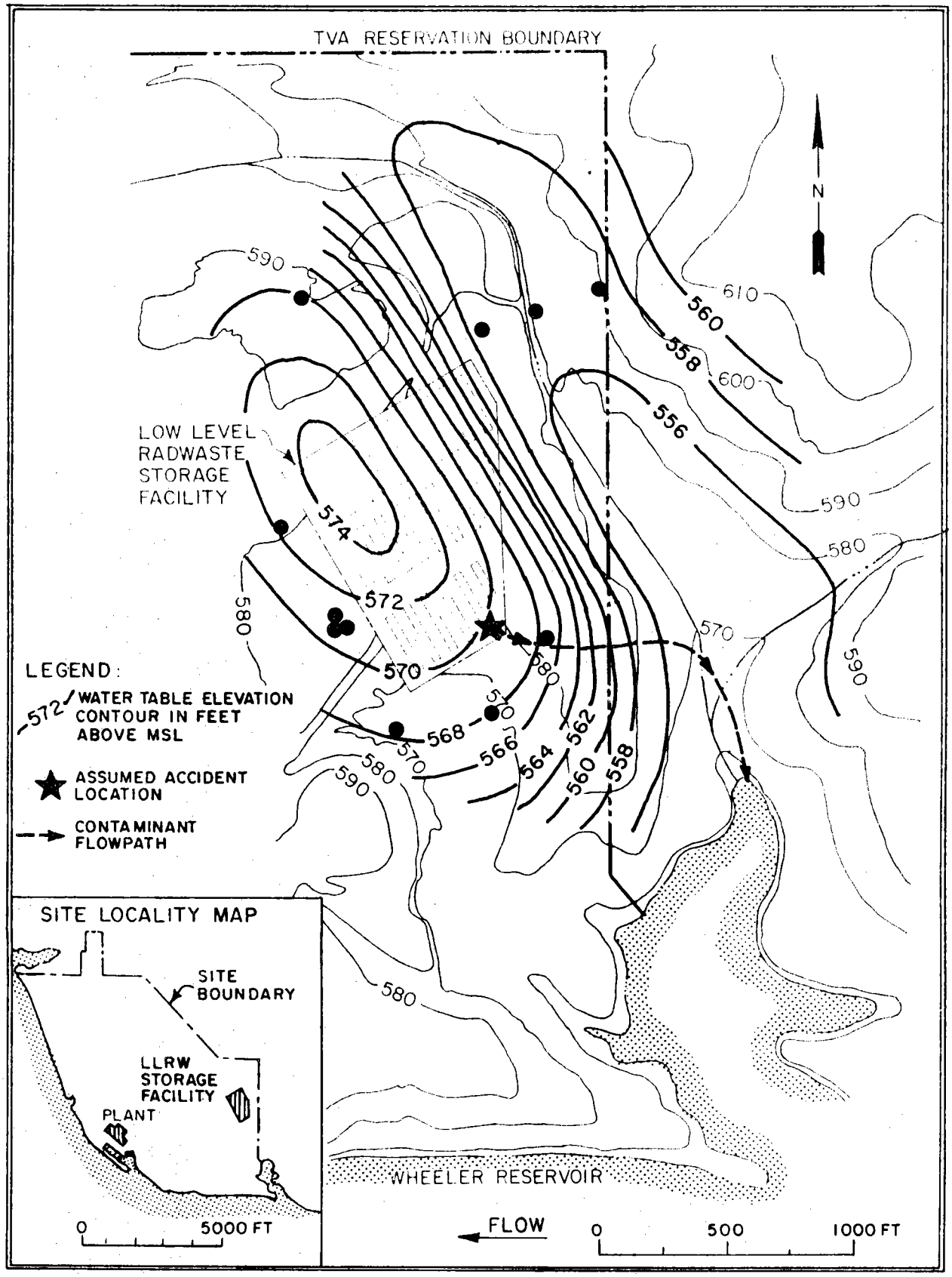
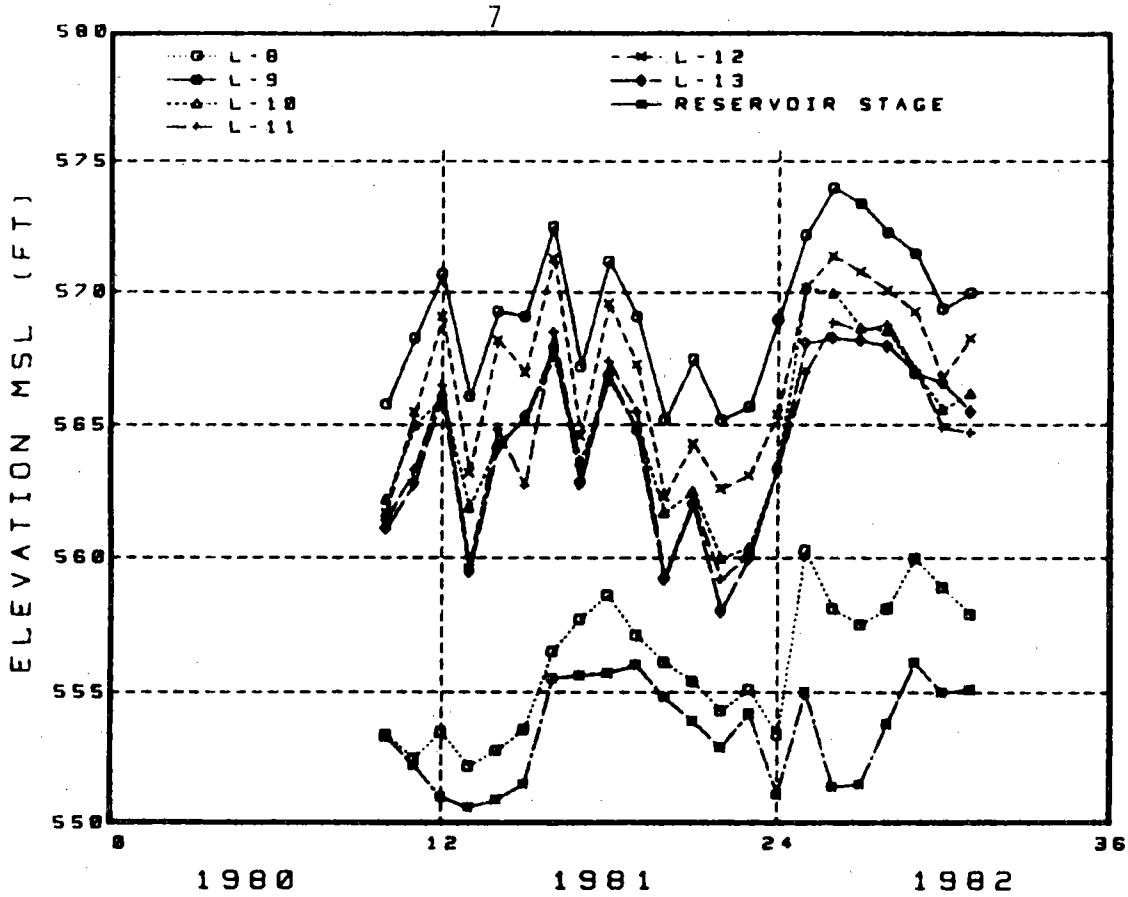


FIGURE 2: Water Table Contour Map



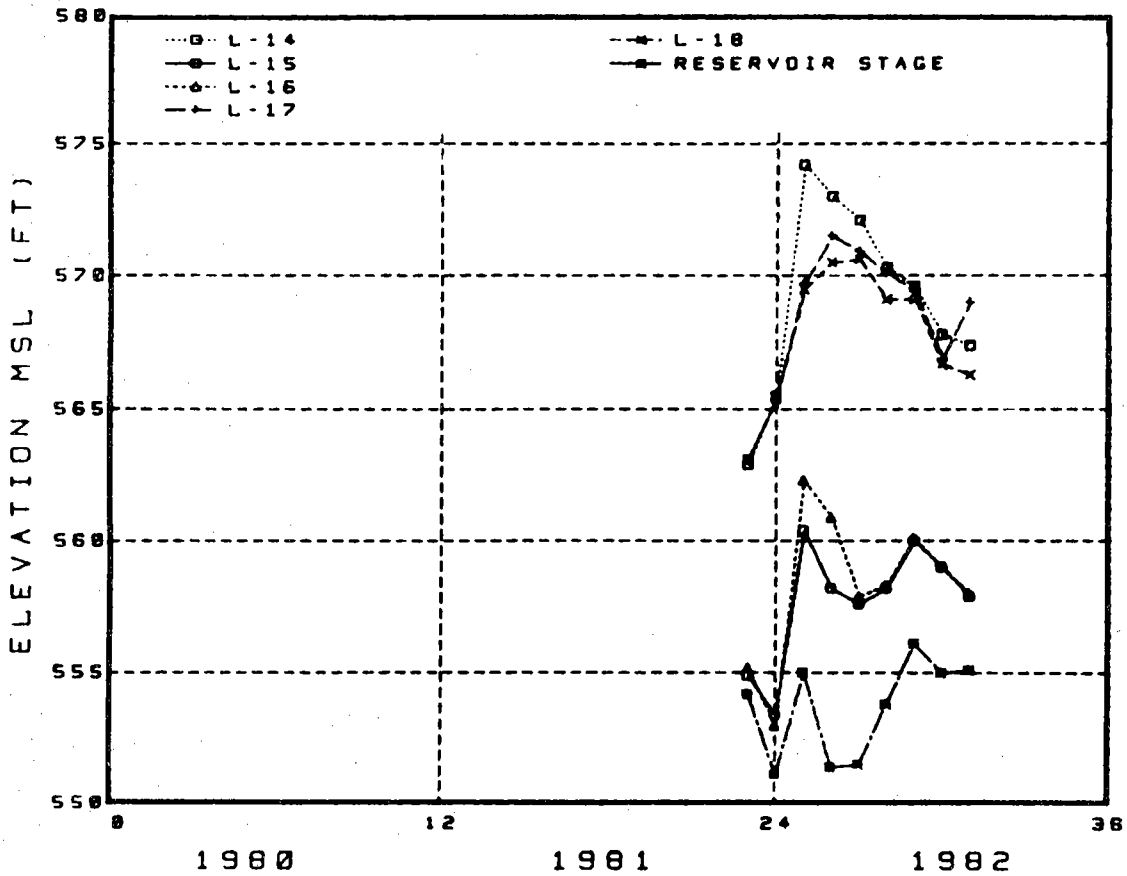


FIGURE 3: Hydrographs for Monitor Wells and Wheeler Reservoir

correlate closely with reservoir fluctuations. They exhibit trends which are markedly different from other site wells which respond more closely with precipitation events. These data suggest that the three wells located on the drainage trend are in close hydraulic communication with the reservoir, most likely via bedrock solution cavities.

Ground-water recharge at the site is derived from infiltrating precipitation. Annual precipitation averages approximately 52 inches. Based on regional water balance studies approximately 10 to 13 inches infiltrates into the ground-water system annually (Zurawski, 1978). All ground water originating at the LLRWSF eventually discharges into the Tennessee River.

ACCIDENT SCENARIO

The accident considered in the analysis is believed to be a worst-case event having a very low probability of occurrence. It is assumed that one percent of the maximum 5-year total activity (corresponding to the term of the pending NRC license) is released per year in liquid form through a crack in the floor of the storage module indicated in Figure 2. The dissolved nuclides enter the ground-water system beneath the module foundation and ultimately discharge into the Tennessee River. The release is assumed to occur over a 5-year period since specific waste items will only be stored in modules for a maximum of 5 years before being removed for permanent disposal elsewhere. During this period, the contaminant source materials as well as the migrating nuclides decay according to their appropriate nuclear decay rates. Five radionuclides were evaluated in the analysis: Sr-90, Cs-137, Cs-134, Co-60, and Mn-54. Characteristics of the source contaminants used in the analysis are presented in Table 2.

TABLE 2: Characteristics of Radiological Contaminants

Nuclide	Release Rate (Ci/yr)	Initial Concentration (μCi/ml)	Decay Constant (day^{-1})	Half Life (yrs)
Sr-90	1.00E-01	9.42E-02	6.62E-05	28.6
Cs-137	4.25E+02	4.00E+02	6.30E-05	30.1
Cs-134	1.75E+01	1.65E+01	8.25E-04	2.3
Co-60	4.00E+00	3.77E+00	3.54E-04	5.4
Mn-54	7.00E-01	6.59E-01	2.19E-03	0.9

The storage module indicated in Figure 2 represents a worst-case accident location because of its close proximity to both the site boundary and the reservoir. It was selected for evaluaton before the decision to reduce the number of storage modules and is not a part of the current facility design. Nevertheless, results are still applicable to the extent that the impacts associated with an accident at the currently planned LLRWSF would be even less than those for the accident location assumed in this analysis due to the longer ground-water flowpath.

MODELING ANALYSIS

The ground-water flow field in the vicinity of the LLRWSF is complex from both the areal and vertical viewpoints. However, some simplification of the dimensionality of the problem is necessary since the expense involved in a three-dimensional analysis is not justified in this case. The strong vertical component of transport within the soil followed by predominantly horizontal movement in the bedrock requires use of a two-dimensional vertical profile model. The profile model is applied along the ground-water streamline indicated in Figure 2. This flowpath represents a worst-case accident condition in terms of minimum distance and contaminant travel time to the site property boundary and to the reservoir.

Modeling of the ground-water transport of dissolved nuclides along the assumed flowpath is conducted in two steps. The first step involves development and calibration of a hydrodynamic model with which to determine the ground-water velocity field. The velocity field is incorporated into a ground-water solute transport model in the second step to determine the temporal and spatial distribution of contaminants as they disperse in the ground-water system.

Hydrodynamic Model

A two-dimensional finite-element hydrodynamic model developed by Yeh and Ward (1979) was used to obtain the ground-water velocity field. The finite element mesh and boundary conditions are shown in Figure 4. The left-hand boundary is located at the approximate center of the ground-water table mound in the LLRWSF area, and, therefore, a no-flow condition is specified at this boundary of the model. The right-hand model boundary is located at the reservoir's edge. Constant hydrostatic head conditions are specified along this boundary. Radionuclide transport is assumed to occur within the soil zone and within approximately the upper 100 feet of bedrock. A no-flow condition is assumed along the lower model boundary. Constant recharge flux is assumed across the upper model boundary. Initially a uniform recharge flux of 2.6E-8 ft/sec was applied to this boundary in accordance with the estimated average annual ground-water recharge rate of 10 inches per year for the site region.

An initial hydraulic conductivity of 3.7E-08 ft/sec was assigned to the soil zone based upon the average measured value (Table 1).

Note that the saturated hydraulic conductivity for the soil was applied uniformly to both the saturated and partially saturated soil zones in

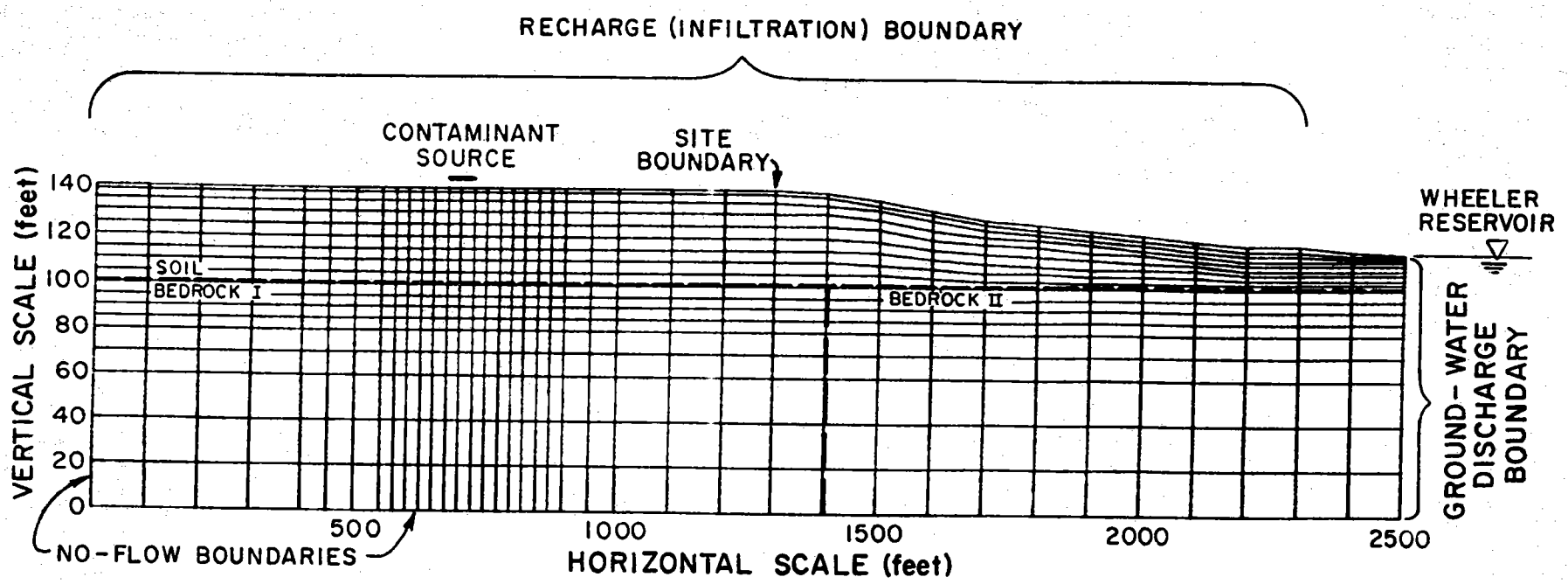


FIGURE 4: Finite Element Grid for Hydrodynamic Model

the modeled region. Under natural conditions site soils are close to being completely saturated, e.g., the average natural saturation level is 93 percent (Table 1). The partially saturated conductivity at this level of saturation would be only slightly less than the natural conductivity. Therefore, the more conservative saturated value of hydraulic conductivity is assumed for the partially saturated as well as fully saturated soils. The bedrock was divided into two zones having different hydraulic characteristics consistent with the geohydrologic inferences discussed previously. The region of minor bedrock solution cavity development is designated as zone I and is assigned a hydraulic conductivity of 2.3E-05 ft/sec on the basis of regional transmissivity values (Moran, 1977). Bedrock zone II is the region where extensive solution cavity development is inferred and a conductivity of 2.3E-04 ft/sec is specified in this region of the model.

The hydrodynamic model was calibrated by adjusting the input parameters--particularly those parameters in which there was the least amount of confidence--until a reasonably close match was obtained between the computed and observed water table positions as shown in Figure 5. In the calibration process the soil hydraulic conductivity was increased to 1.3E-07 ft/sec. This is a reasonable and conservative adjustment considering that laboratory measured conductivities tend to be lower than actual values. Bedrock hydraulic conductivities were held at their original values, as the model was found to be relatively insensitive to these parameters. It was necessary to separate the upper recharge boundary of the model into two areas. Recharge rates of 2.4E-08 ft/sec (9 ins/yr) along the horizontal portion of the boundary and 1.3E-08 ft/sec (5 ins/yr) where the ground surface slopes toward the reservoir provided the best calibration results.

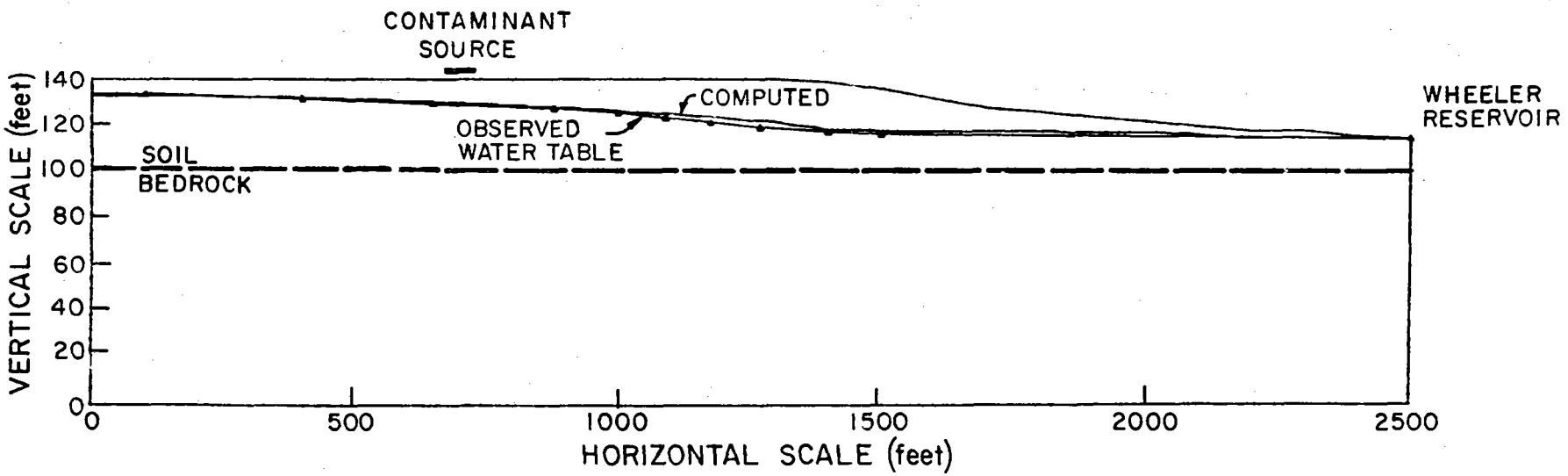


FIGURE 5: Comparison of Computed and Observed Water Table

The Darcian velocity distribution for the calibrated model is shown in Figure 6. The figure indicates that the velocities in the soil are small and generally directed vertically downward. At the soil-rock interface the velocities increase and the direction changes abruptly, becoming nearly horizontal in the rock. The sharp refraction of ground-water velocity at the soil-rock interface is due to the large difference between the hydraulic conductivities of the soil and bedrock.

Contaminant Transport Model

A two-dimensional ground-water solute transport model developed by ERAS, Inc. (1982) was used to predict the migration and fate of radiological contaminants released during the postulated accident. The model known as "WASTER" uses the discrete element method to solve the following equations governing the transport of a solute subject to linear equilibrium adsorption and first-order decay:

$$\theta R_f \frac{\partial c}{\partial t} = \frac{\partial}{\partial x}\left(D_{xx}\frac{\partial c}{\partial x} + D_{xz}\frac{\partial c}{\partial z}\right) + \frac{\partial}{\partial z}\left(D_{zz}\frac{\partial c}{\partial z} + D_{zx}\frac{\partial c}{\partial x}\right) - \frac{\partial q_x c}{\partial x} - \frac{\partial q_z c}{\partial z} - \lambda c \qquad (1)$$

where, c = solute concentration $[M/L^3]$

D_{xx}, D_{xz}, D_{zx}, D_{zz} = hydrodynamic dispersion coefficients $[L^2/T]$

q_x, q_z = Darcian velocity components $[L/T]$

R_f = retardation factor = $1+\rho K_D/\theta$ [dimensionless]

K_D = distribution coefficient $[L^3/M]$

x, z = horizontal and vertical coordinates, respectively $[L]$

t = time $[T]$

λ = decay constant $[L^{-1}]$

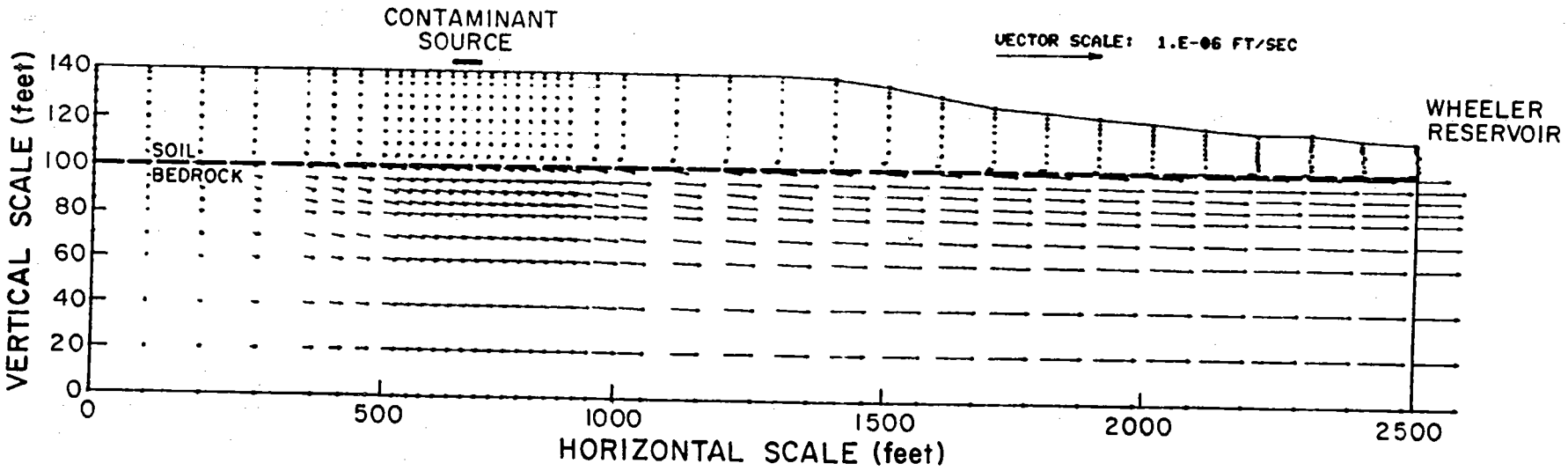


FIGURE 6: Ground-Water Velocity Field

ρ = bulk density of soil/rock [M/L^3]

θ = soil/rock moisture content [dimensionless]

The dispersion coefficients are related to media properties and flow field as follows:

$$D_{xx} = \alpha_L \, q_x^2 / \bar{q} + \alpha_T \, q_z^2 / \bar{q} \tag{2}$$

$$D_{xz} = D_{zx} = (\alpha_L - \alpha_T) \, q_x q_z / \bar{q} \tag{3}$$

$$D_{zz} = x_L \, q_z^2 / \bar{q} + \alpha_T \, q_x^2 / \bar{q} \tag{4}$$

where $\bar{q} = (q_x^2 + q_z^2)^{\frac{1}{2}}$

α_L = longitudinal dispersivity [L]

α_T = transversal dispersivity [L]

The element mesh for the integrated finite difference model is similar to the finite element mesh shown in Figure 4. The left-hand and lower boundaries of the transport model are no-flow boundaries consistent with the hydrodynamic model. The right-hand model boundary is an open boundary through which contaminated ground water ultimately discharges. Two different boundary conditions are specified on the upper model boundary. In the small region of the contaminant source it is assumed that

$$c = \begin{cases} c_o \exp(-\lambda t) \;, & 0 \leqq t \leqq 5 \text{ years} \\ 0 \;, & t > 5 \text{ years} \end{cases} \tag{5}$$

where c_o is the initial radionuclide concentration. Across the remainder of the upper boundary, $c = 0$ for $t \geqq 0$ to simulate recharge of uncontaminated precipitation. The initial contaminant concentration is assumed to be zero throughout the modeled ground-water system except at the contaminant source.

The steady-state velocity distribution (Figure 6) obtained from the hydrodynamic model is used in calculation of the convective and dispersive transport of contaminants. The dispersivity of the soil and bedrock

were not measured due to the great expense involved in these measurements. In place of measured data, the soil and bedrock dispersivities shown in Table 3 have been assumed for the analysis. These values are based upon measured values for similar geologic materials reported in the literature (Anderson, 1979; Gelhar and Axness, 1981).

TABLE 3: Assumed Range of Soil/Bedrock Dispersivities

	Minimum α_L/α_T(ft)	Expected α_L/α_T(ft)	Maximum α_L/α_T(ft)
Soil	0.5/0.25	2.0/1.0	4.0/2.0
Bedrock, Zone I	10./1.	50./5.	100./10.
Bedrock, Zone II	10./1.	50./5.	100./10.

The sorptive characteristics of the soils beneath the LLRWSF area were determined through laboratory testing of core samples (Rogers, 1982). Batch techniques were used on composite soil samples to measure distribution coefficients (K_D) for Cs-137, Cs-134, Sr-90, Co-60, and Mn-54. Laboratory results are given in Table 4. The sorptive capacity of the bedrock was not measured, but is believed to be negligible. Therefore, conservative K_D values of zero were assumed in all cases for bedrock.

Results

Model simulations were conducted to determine the maximum radionuclide concentrations resulting from the postulated accident at two possible receptors: a hypothetical water supply well located near the site boundary and Wheeler Reservoir. Because of the uncertainties associated with some model input data it was necessary to perform a parametric analysis in order to bracket the possible accident impacts. The parametric analysis indicates

TABLE 4: Soil Distribution Coefficients (K_D)

Sample No.	Sr-90 (ml/gm)	Cs-137 Cs-134 (ml/gm)	Co-60 (ml/gm)	Mn-54 (ml/gm)
1	90 (10)	3600 (200)	410 (20)	150 (10)
2	70 (10)	6400 (400)	460 (20)	160 (10)
3	90 (10)	8600 (600)	480 (20)	160 (10)
4	71 (3)	--	1080 (50)	237 (6)
5	72 (3)	--	1190 (50)	223 (6)
6	69 (3)	--	1030 (50)	211 (5)
7	116 (6)	--	--	--
8	107 (6)	--	--	--
9	109 (6)	--	--	--

Notes:
1) K_D data obtained from Table 3 of Rogers (1982).
2) Numbers in parentheses ( ) represent experimental error estimates.

how the maximum contaminant concentrations at the receptors are affected when individual parameters are varied over a range of potential values. The parameters investigated include soil and bedrock dispersivities, soil K_D, net ground-water recharge rate, and bedrock porosity. Other parameters were not investigated either because they had been adequately measured or because they were known to have little effect on model results.

The cost of computer simulations involved in parametric analyses on the above five parameters for each radionuclide would be prohibitive. In order to minimize computer costs, simulations were made only for Sr-90. In general for a given set of geohydrologic conditions the maximum relative concentration (the absolute concentration divided by the initial concentration or c/c_0) achieved by each radionuclide depends only on its decay rate and

distribution coefficient. The lower the magnitude of either parameter, the higher the relative concentration. Compared with the other nuclides, Sr-90 attains the highest relative concentration at the receptors because of its relatively slow decay rate and low attenuation by soils. Once the maximum relative concentration of Sr-90 is calculated conservative estimates of the maximum absolute concentrations of Cs-137, Cs-134, Co-60, and Mn-54 are computed by multiplying the initial concentration of each nuclide by the relative concentration of Sr-90. If the resulting absolute concentration for each nuclide is within acceptable limits there is no need to conduct separate modeling analyses for each nuclide.

The "base case" predictive simulation refers to the simulation made using the most probable values for model input parameters. Thus, the base case results represent the expected accidental release impacts. The range of parameter values used in the analysis are listed in Table 5.

The results of the parametric study are presented graphically in Figures 7 through 11. These figures illustrate how the maximum relative concentration values for Sr-90 at the site boundary and reservoir vary with changes in each parameter while other model parameters are held at their base case values. In all cases the initial contaminant concentrations have been reduced by a factor of at least 10^{-15} at the potential receptors. Note that the calculated contaminant concentrations entering the reservoir would be reduced further by mixing within the reservoir; however, no credit is taken for this additional attenuation. The predicted relative and actual contaminant concentrations under base case and worst case assumptions are given in Table 6. In all cases the actual contaminant levels are well below 10 CFR Part 20 limits.

TABLE 5: Model Input Parameters

Parameter	Minimum	Base Case	Maximum
Sr-90 K_D, soil (ml/gm)	69	88	116
Sr-90 K_D, bedrock I/II (ml/gm)	0	0	0
α_L/α_T, soil (ft)	0.5/0.25	2.0/1.0	4.0/2.0
α_L/α_T, bedrock I/II (ft)	10/1	50/5	100/10
θ, soil	*	0.46	*
θ, bedrock I/II	0.05	0.10	0.20
ρ, soil (lb/ft^3)	*	92	*
ρ, bedrock I/II (lb/ft^3)	--	--	--
K, soil (ft/sec)	*	1.3E-07	*
K, bedrock I (ft/sec)	*	2.3E-05	*
K, bedrock II (ft/sec)	*	2.3E-04	*
Net recharge (ft/sec)	1.6E-08/0.8E-08	2.4E-08/1.2E-08	3.2E-08/1.6E-08

*Not evaluated
--Irrelevant when bedrock $K_D = 0$

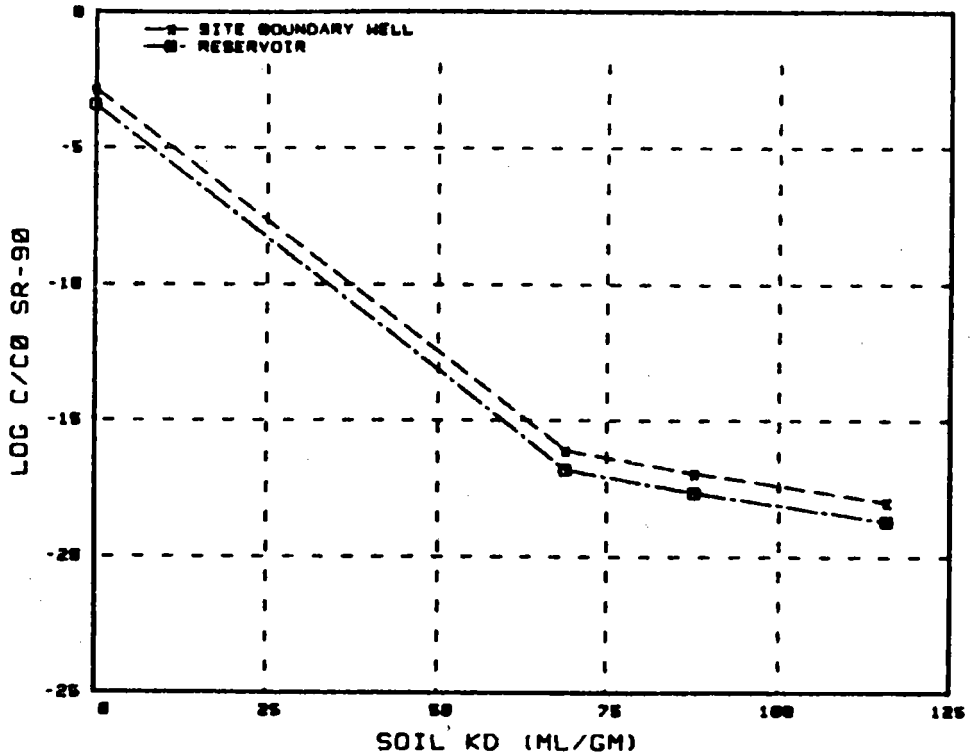


FIGURE 7: Variation of Sr-90 c/c_o with Changes in Soil K_D

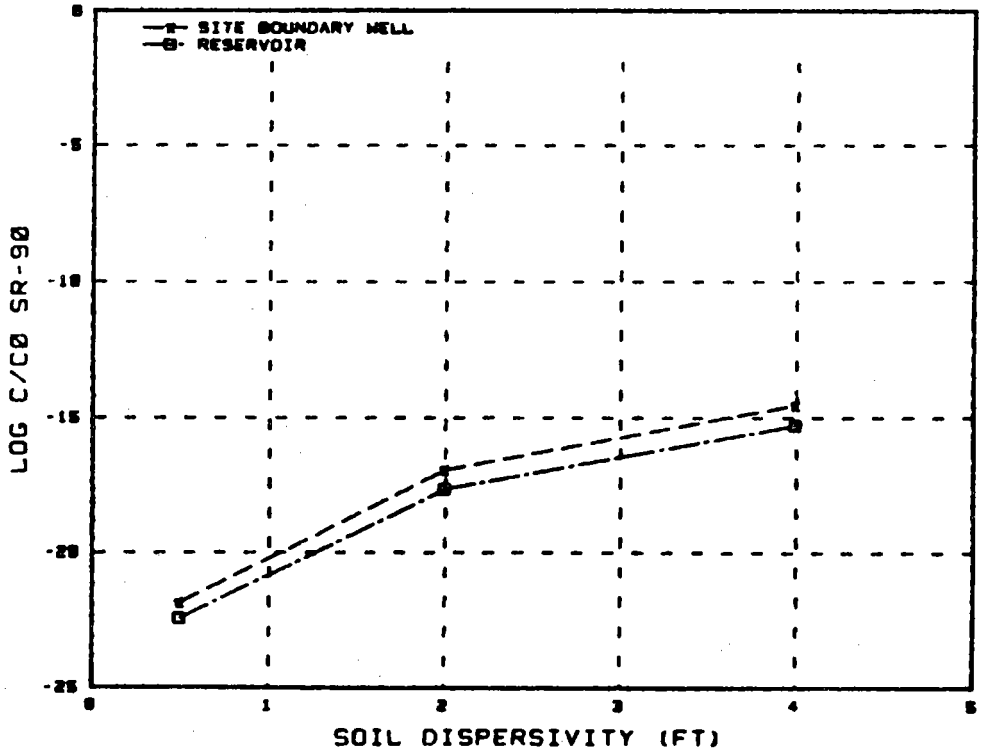


FIGURE 8: Variation of Sr-90 c/c_o with Changes in Soil Dispersivity

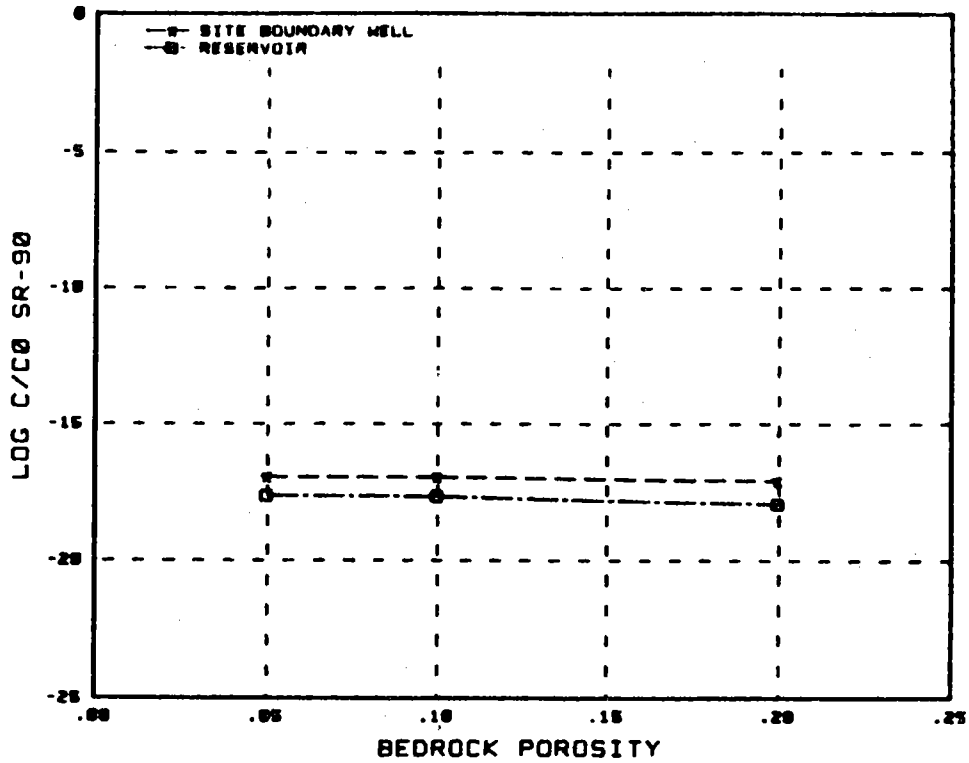


FIGURE 9: Variation of Sr-90 c/c_o with Changes in Bedrock Porosity

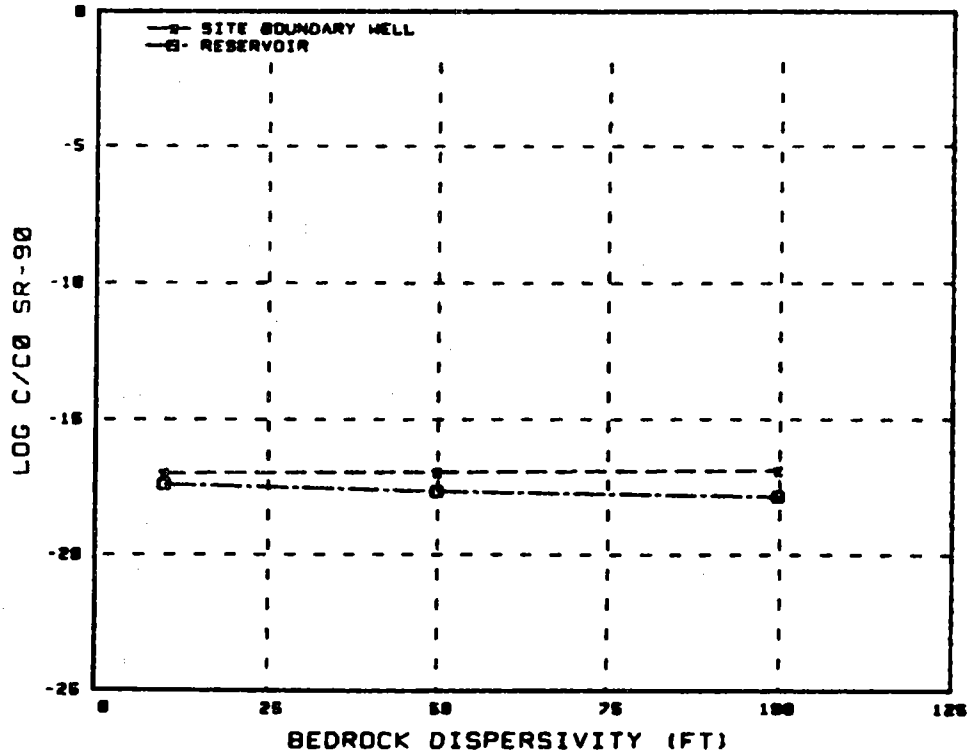


FIGURE 10: Variation of Sr-90 c/c_o Changes in Bedrock Dispersivity

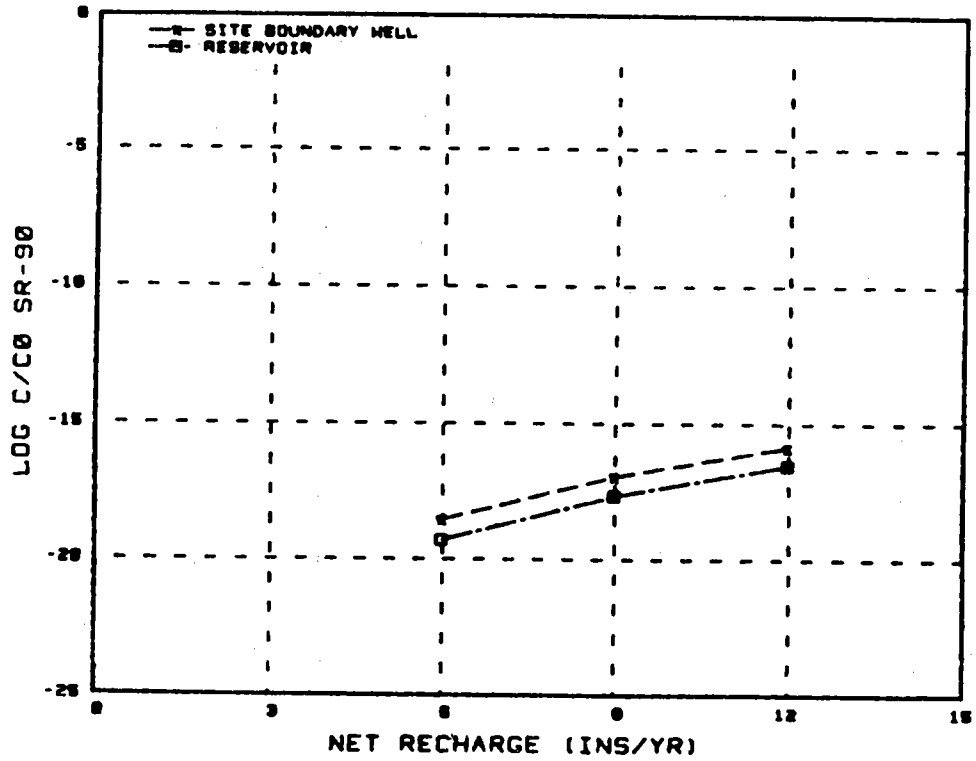


FIGURE 11: Variation of Sr-90 c/c_0 with Changes in Net Recharge

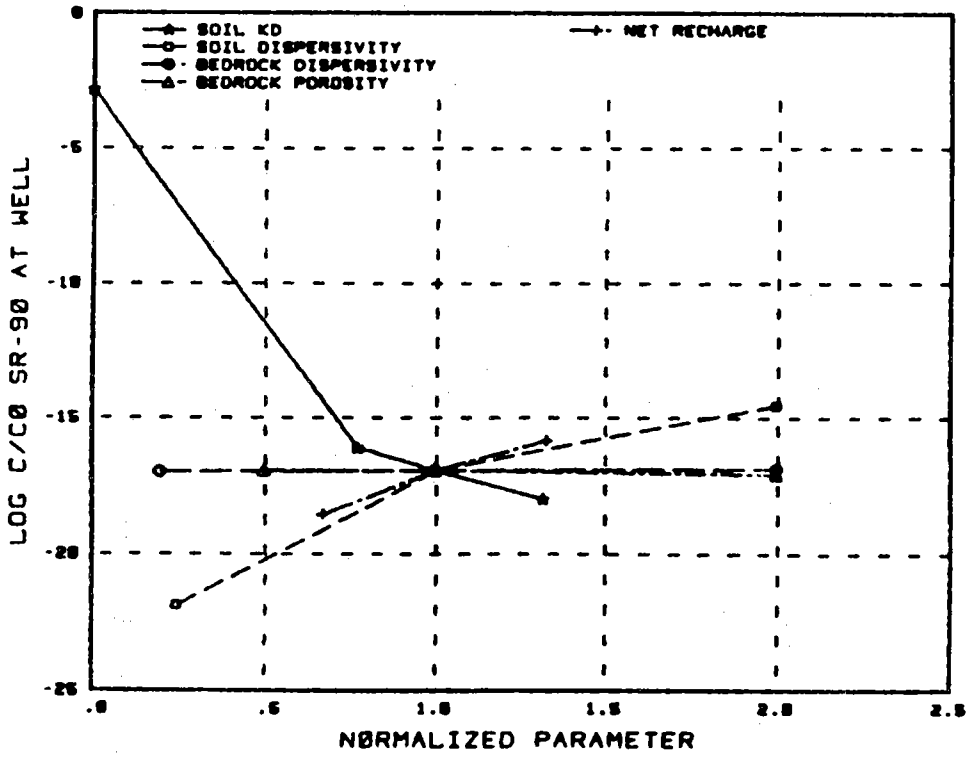


FIGURE 12: Model Sensitivity Analysis Results

TABLE 6: Predicted Radiological Contaminant Concentrations

Radionuclide	Base Case				Worst Case*				10 CFR 20 Limits
	Site Boundary Well		Reservoir		Site Boundary Well		Reservoir		
	c/c_o	c(μci/ml)	c/c_o	c(μci/ml)	c/c_o	c(μci/ml)	c/c_o	c(μci/ml)	(μci/ml)
Sr-90	1.1E-17	1.0E-18	2.1E-18	2.0E-19	2.9E-15	2.7E-16	4.9E-16	4.6E-17	3.0E-07
Cs-137	<1.1E-17	<1.0E-15	<2.1E-18	<8.4E-16	<2.9E-15	<1.2E-12	<4.9E-16	<2.0E-13	2.0E-05
Cs-134	<1.1E-17	<1.8E-16	<2.1E-18	<3.5E-17	<2.9E-15	<4.8E-14	<4.9E-16	<8.1E-15	9.0E-06
Co-60	<1.1E-17	<4.1E-17	<2.1E-18	<7.9-18	<2.9E-15	<1.8E-15	<4.9E-16	<1.8E-15	3.0E-05
Mn-54	<1.1E-17	<7.2E-18	<2.1E-18	<1.4E-18	<2.9E-15	<1.9E-15	<4.9E-16	<3.2E-16	1.0E-04

*Worst-case results were generated by parametric run in which soil α_L/α_T = 4/2 ft.

Appendix A contains graphs of Sr-90 c/c_o versus time at the site boundary and reservoir for each parametric run. In general, these curves indicate that peak concentrations are not achieved at receptors until 300 to 400 years after the initial release.

Results of the parametric analyses are presented in a slightly different form in Figure 12 using normalized parameter values (i.e., parameters that have been normalized by dividing by the base case parameter value). This method of presentation allows comparison of the sensitivity of the model output to changes in one input parameter, with model sensitivity due to changes in other parameters. In general the steeper the slope of the c/c_o versus normalized parameter curve, the more sensitive the model results are to parameter changes, and hence the greater the need for accurate parameter determination. Figure 12 indicates model results are relatively sensitive to changes those parameters which control rate of contaminant transport through the soil, i.e., K_D, soil dispersivity and net recharge rate. On the other hand, the model is insensitive to changes in bedrock dispersivity and porosity which control contaminant movement through the bedrock.

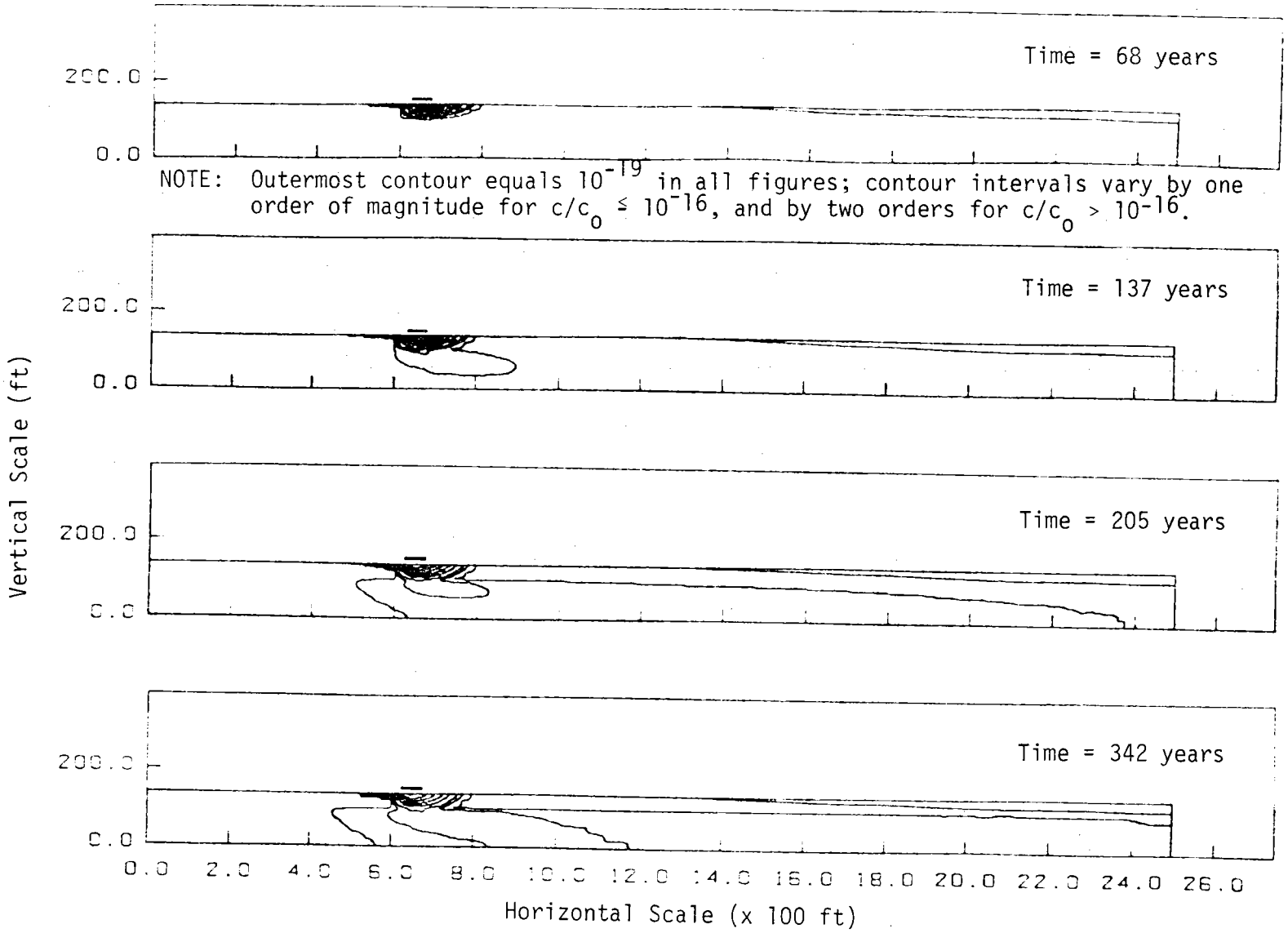
These results provide some insight into the credibility of the model predictions. Soil K_D appears to be the most critical input parameter. The fact that soil distribution coefficients were carefully measured input parameters enhances the confidence which may be placed in the model predictions. Soil dispersivity and net ground-water recharge are both somewhat sensitive model parameters. Although site-specific measurements were not made for either parameter, the base case and extreme values used in the analysis are believed to be reasonable and conservative estimates. Bedrock dispersivity and effective porosity were not measured, but any

error associated with these parameters is of little consequence due to the insensitivity of the model output to these parameters.

On this basis and considering the consistently conservative assumptions used throughout the analysis we conclude that the model predictions represent reliable conservative estimates of the impacts resulting from an accidental release of radwaste contaminants from the LLRWSF.

RECOMMENDATIONS CONCERNING MONITORING AND MITIGATION

In view of present plans to store radwastes only on an emergency basis at onsite facilities, there is no need for long-term ground-water monitoring of the LLRWSF at Browns Ferry. As an alternative to routine ground water monitoring it is recommended that modules be inspected following any period of waste storage. If the inspection reveals a probable subsurface contaminant excursion, a program of ground-water monitoring should then be implemented. The existing network of six monitor wells around the original LLRWSF area can be used to partially meet this need. However, it should be recognized that the present monitor well system provides only limited detection capability for the modified facility design. Based upon the contaminant dispersion pattern shown in Figure 13, the most effective monitoring system would consist of a shallow soil monitor well located near each module and additional bedrock wells located on the eastern side of the modified LLRWSF area. Nevertheless no additional monitor wells are recommended at this time. Due to the slow rate of contaminant movement through the soil there should be ample time to install additional monitor wells (if necessary) in the event of an accident. The quarterly analysis of water samples collected during 1982 and 1983 at the existing



NOTE: Outermost contour equals 10^{-19} in all figures; contour intervals vary by one order of magnitude for $c/c_0 \leq 10^{-16}$, and by two orders for $c/c_0 > 10^{-16}$.

FIGURE 13: Base Case Sr-90 Isoconcentration Graphs

wells should provide adequate background characteristics of ambient ground-water quality for any future comparisons.

The results of the contaminant transport analysis indicates that none of the potential contaminants reach levels which might be considered hazardous, even in the absence of interdictive measures. Therefore, from a technical viewpoint no mitigation should be necessary assuming the quantities and activities of any future release are no greater than those assumed in the analysis. However, if mitigation becomes expedient, for whatever reason, the only practical alternative is removal of contaminated soil. Due to the slow rate of movement of contaminants through the soil it is likely that any excursion detected within five years of occurrence would not have migrated more than a few feet into the soil. Thus, removal of contaminated soil should be feasible and relatively inexpensive. This procedure as with any interdictive procedure should begin with a drilling and sampling program to delineate the extent of contamination. Analysis of ground water from existing monitor wells should also be performed to assure that no contaminated ground water has migrated offsite.

The long-term nature of the contaminant transport process indicated in the analysis precludes containment-type interdictive measures. For example, construction and maintenance of a network of interceptor wells around the release area to remove contaminated water and prevent offsite movement would be extremely expensive and would only be effective in a timeframe of tens to hundreds of years.

CONCLUSIONS

1. Based upon a conservative ground-water pathway analysis the radiological impacts resulting from an accidental release of radwaste con-

taminants from the LLRWSF are estimated to be insignificant. The calculated contaminant concentrations at a hypothetical water supply well located near the eastern site boundary and at Wheeler Reservoir are well below limits established in 10 CFR Part 20.

2. Because the rate of contaminant movement in the soil zone is so much slower than that in bedrock, the overall rate of contaminant transport is governed by factors controlling the rate of movement through the soil zone including the hydraulic and sorptive soil characteristics and the net ground-water recharge rate. This implies that in planning data acquisition programs for future contaminant transport analyses in similar geohydrologic environments, soil properties and ground-water recharge estimates should be emphasized.

3. Long-term ground-water monitoring should not be necessary at the LLRWSF except in the event of a contaminant excursion. The existing monitor well network is not well suited for the current facility design. However, no additional wells are recommended at this time. There would be sufficient time to install additional monitor wells (if necessary) should an accident occur due to the slow rate of contaminant migration through the soil zone.

4. Accident mitigation is technically unwarranted. If mitigation is expedient for other reasons, removal of contaminated soil is judged the most cost-effective method of interdiction.

5. The level of analysis of the present study should not be necessary for evaluating potential LLRWSF accidents at TVA's other nuclear plant sites provided these facilities are used only for emergency storage.

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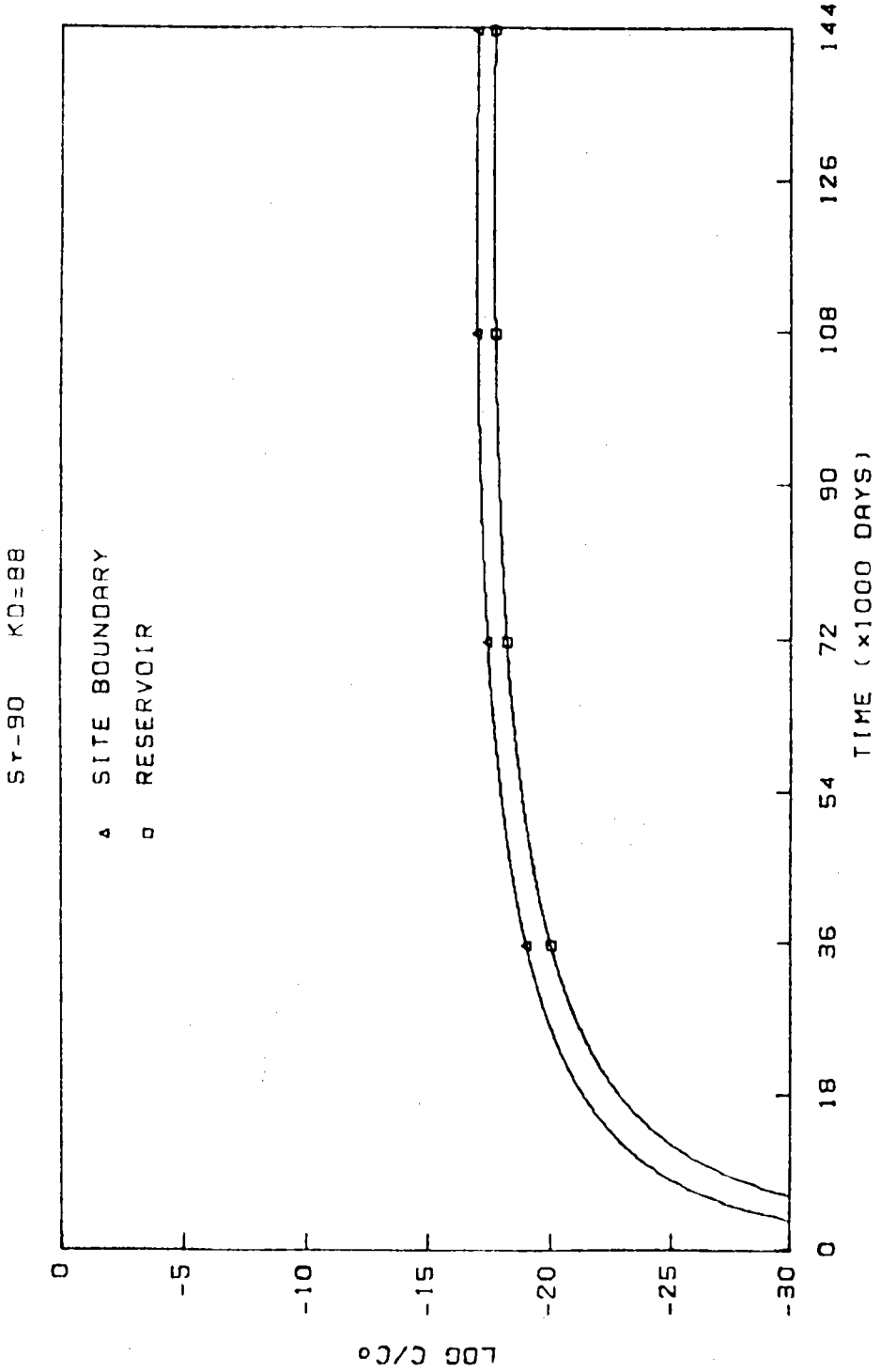
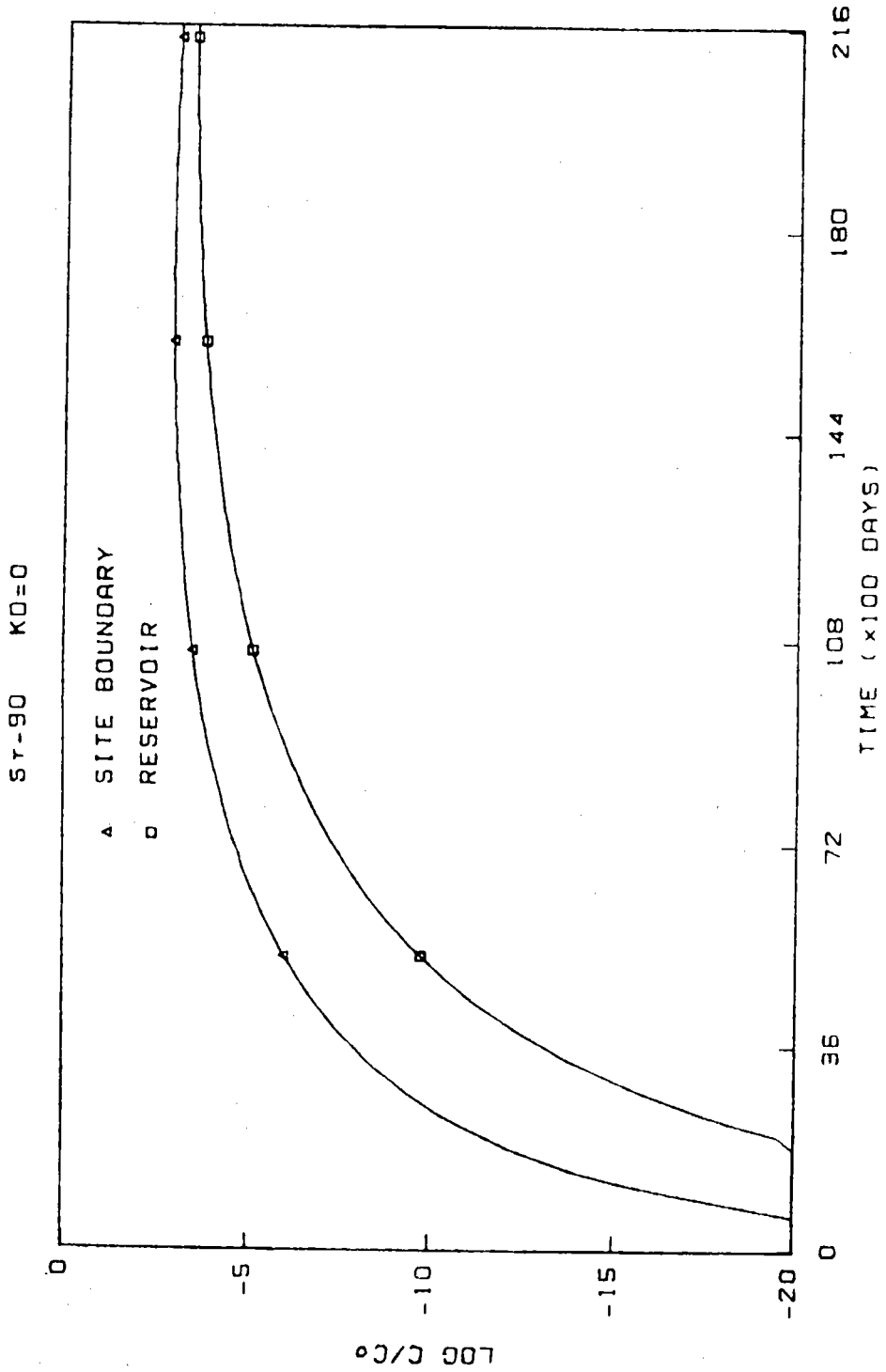
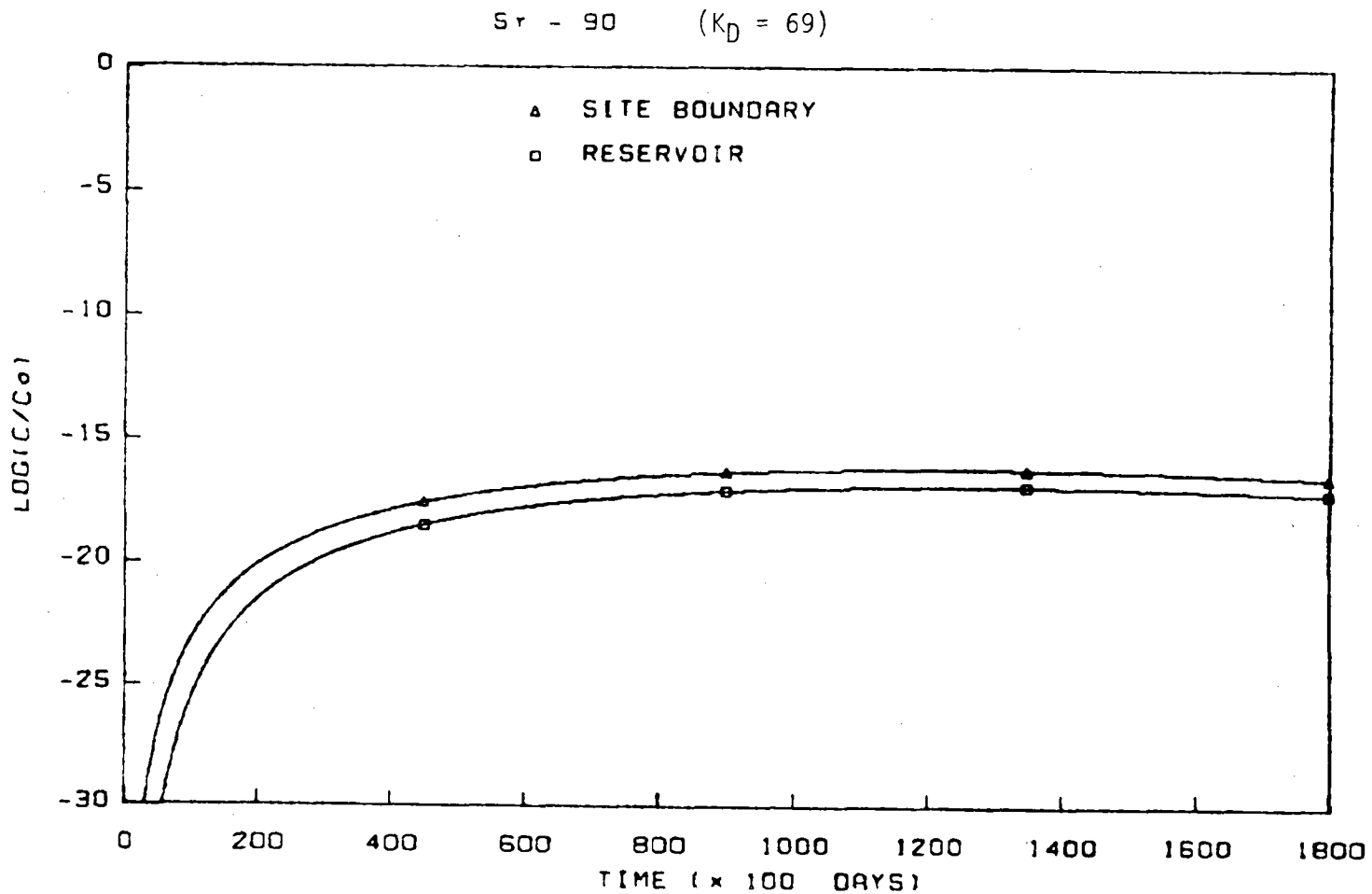
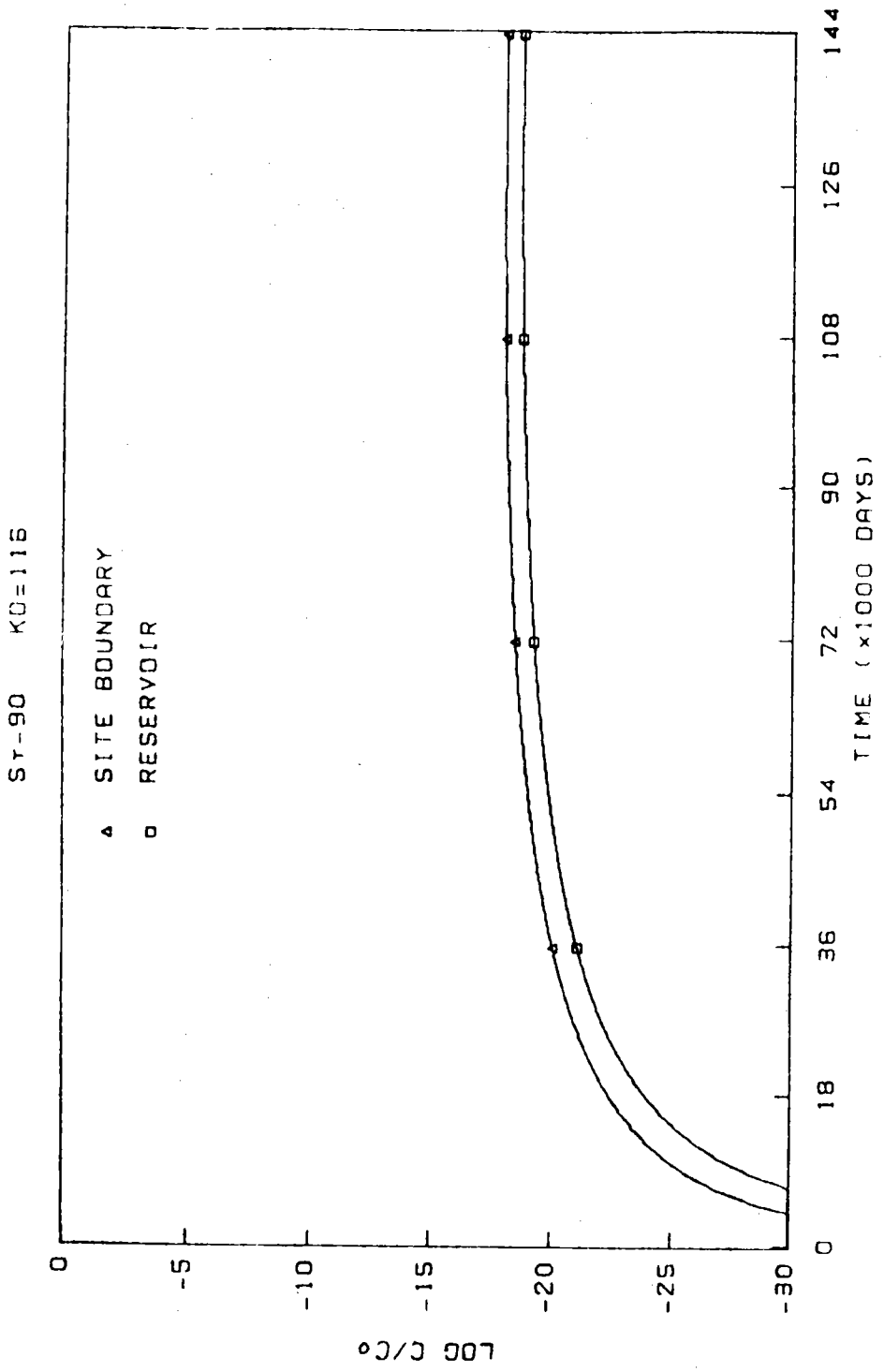
APPENDIX

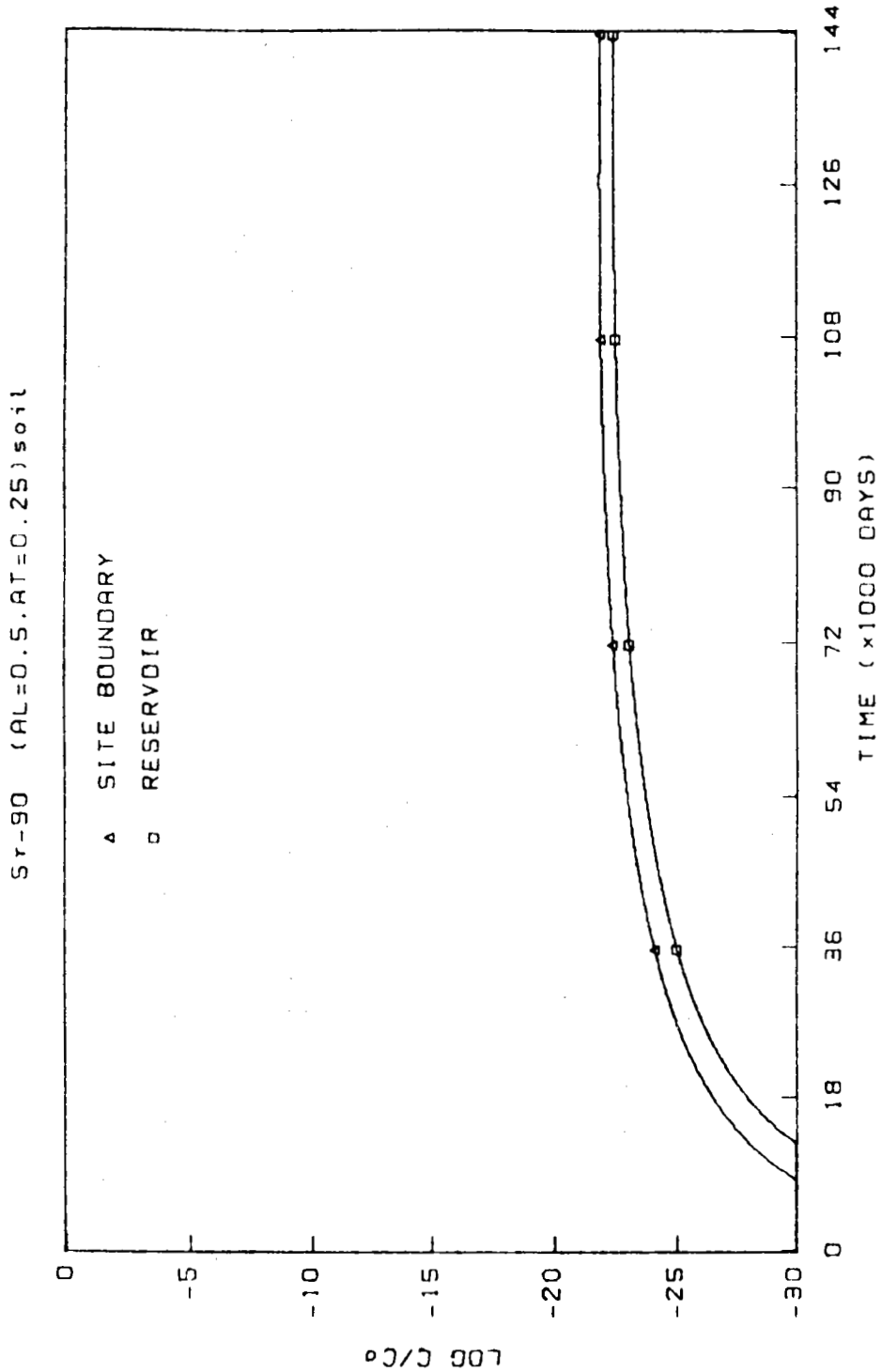
SR-90 C/C_0 VERSUS TIME AT RECEPTORS FOR PARAMETRIC ANALYSIS RUNS

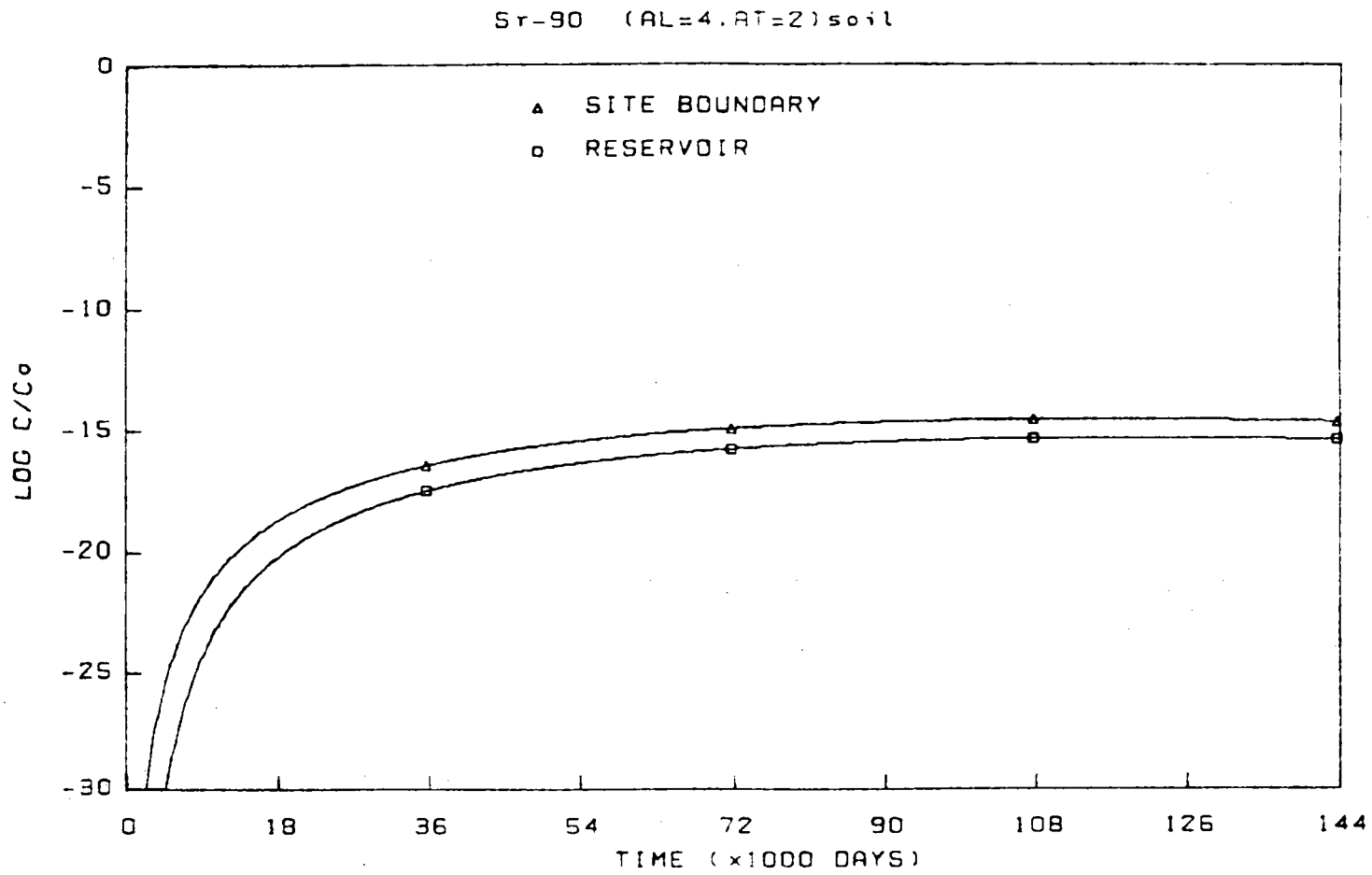

Sr-90 KD=88
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
TIME (x1000 DAYS)
0
18
36
54
72
90
108
126
144

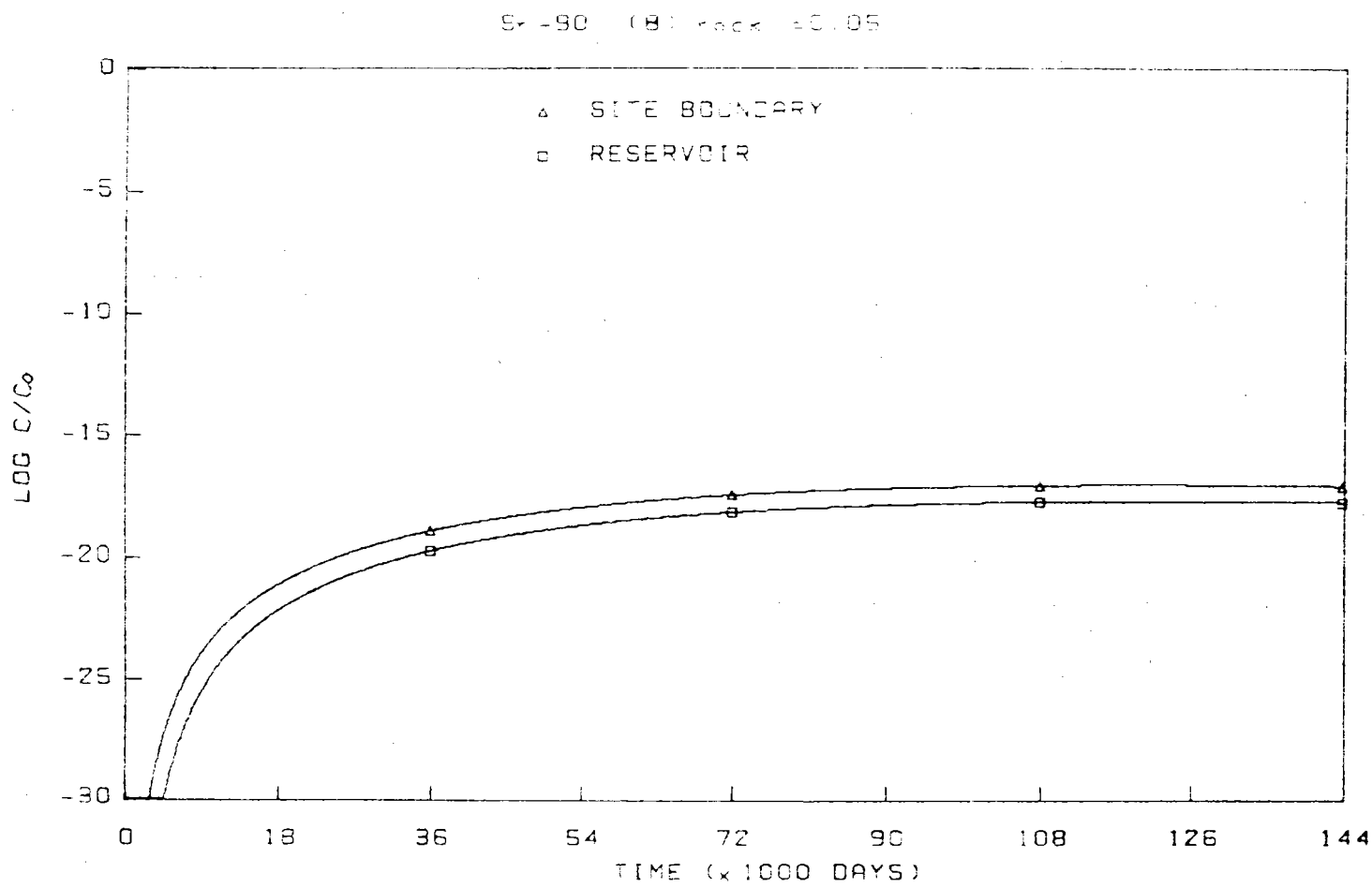

Sr-90 KD=0
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
0
36
72
108
144
180
216
TIME (x100 DAYS)

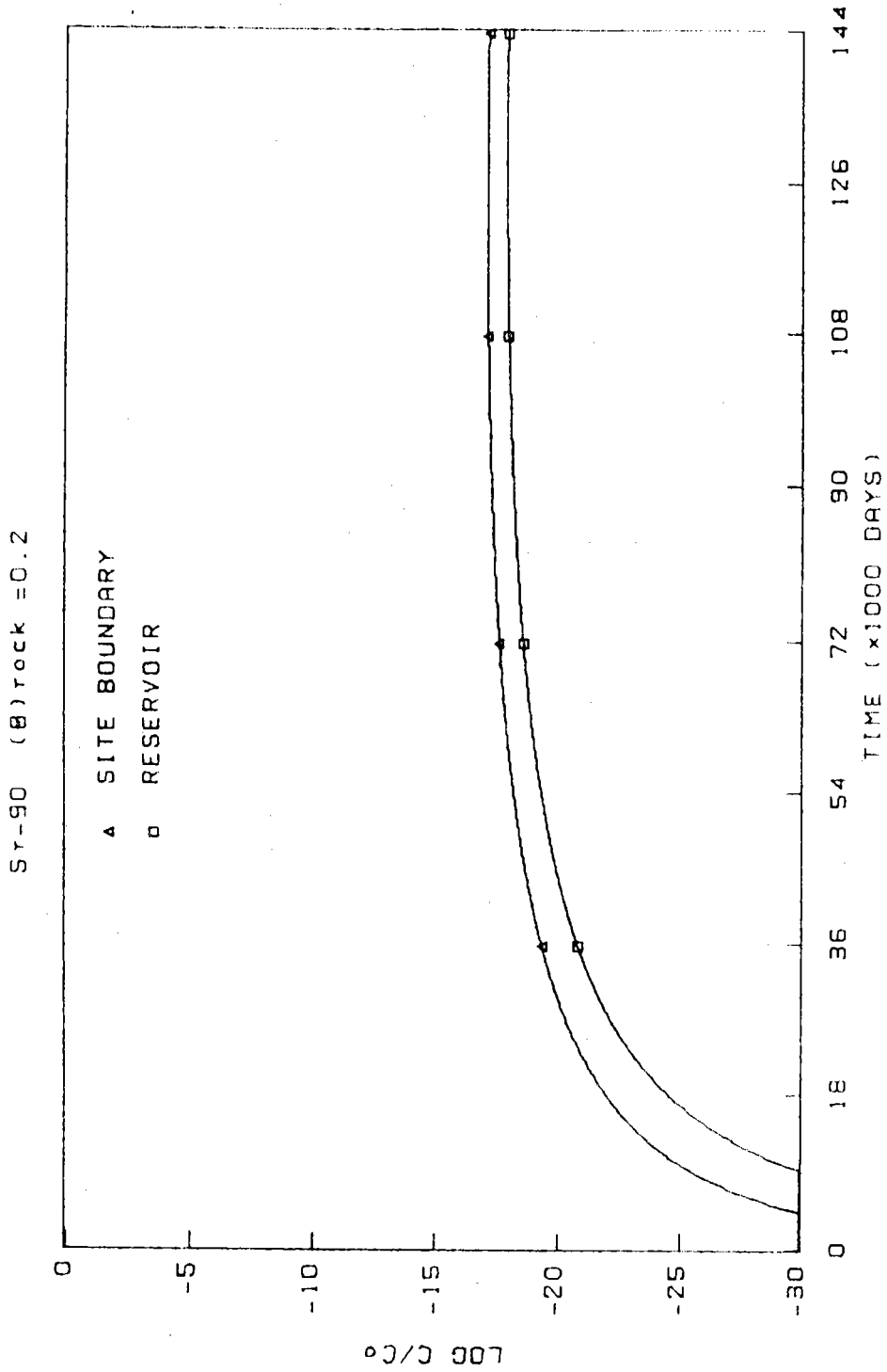

Sr - 90 (K_D = 69)
SITE BOUNDARY
RESERVOIR
LOG(C/Co)
0
-5
-10
-15
-20
-25
-30
0
200
400
600
800
1000
1200
1400
1600
1800
TIME (x 100 DAYS)

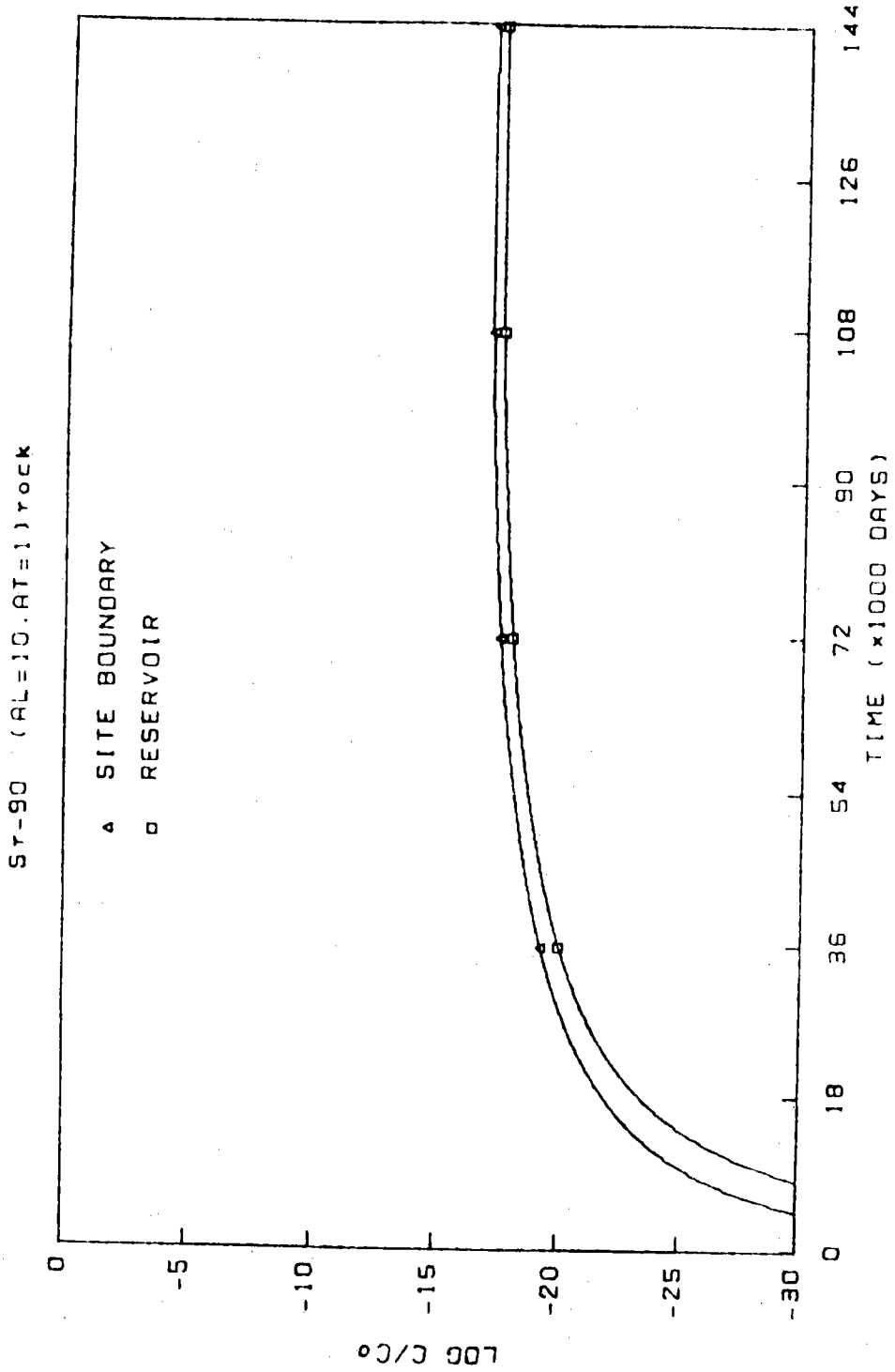

Sr-90 KD=115
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)

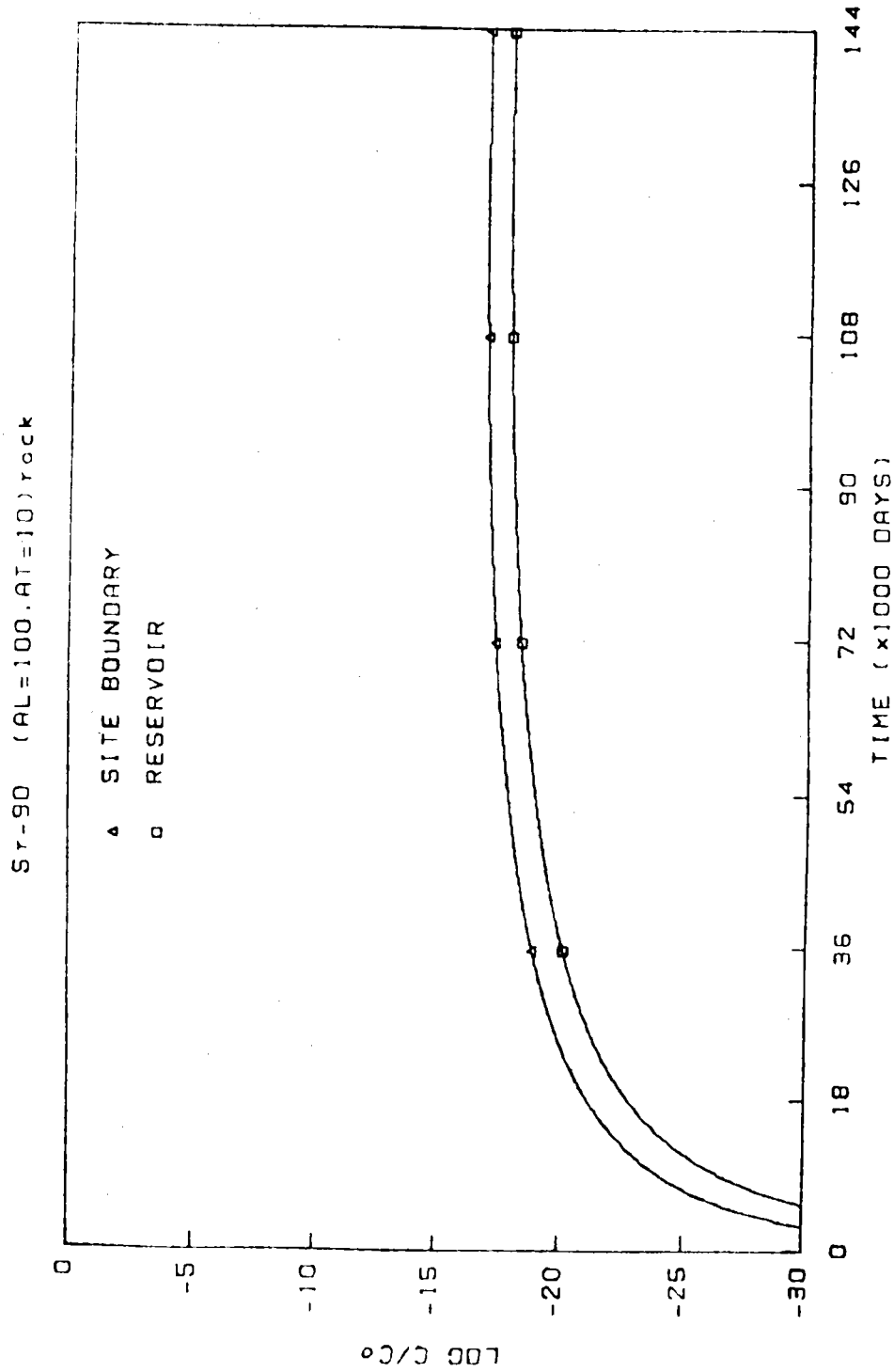

Sr-90 (AL=0.5,AT=0.25)soil
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)

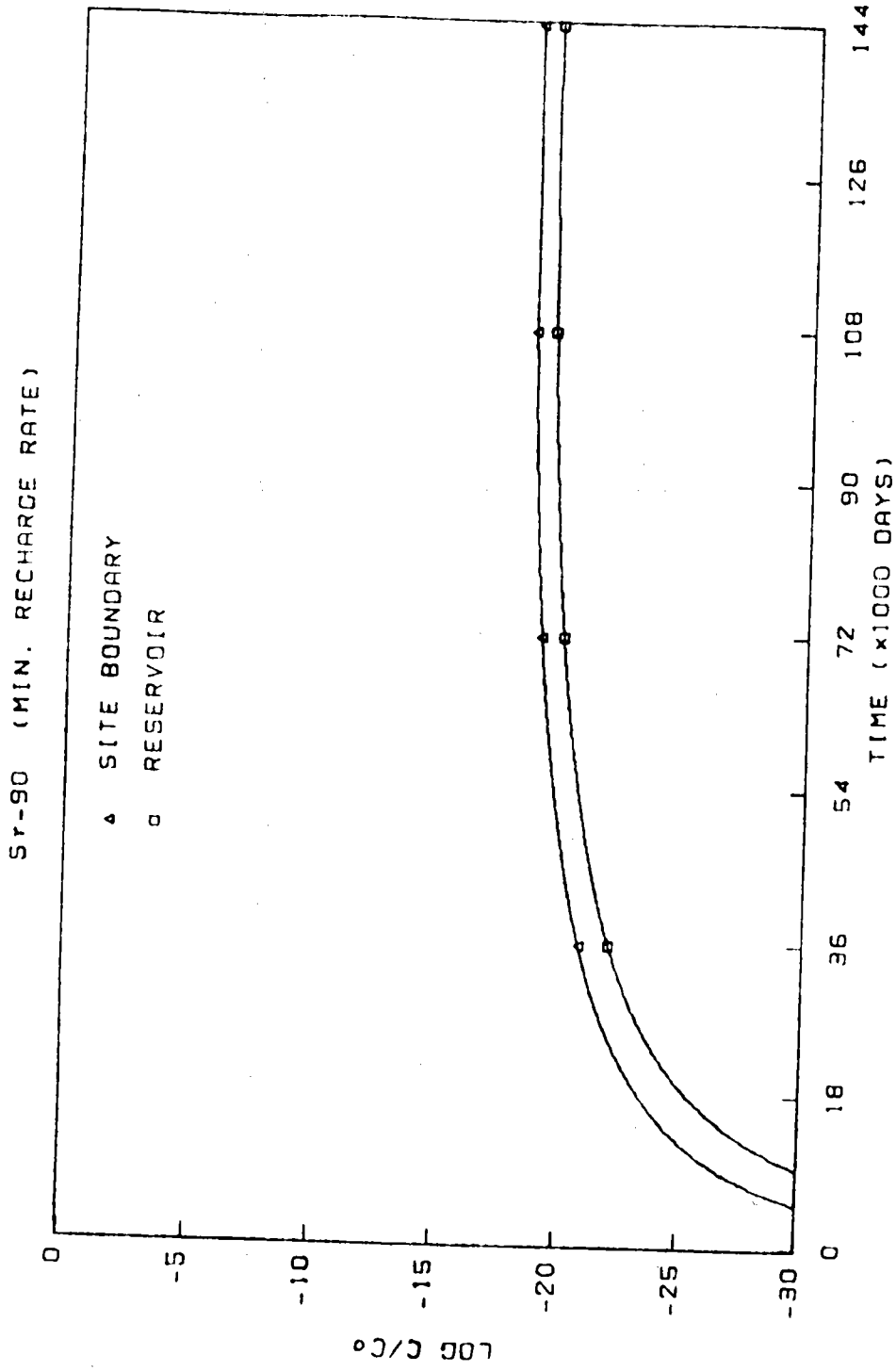

Sr-90 (AL=4.AT=2)soil
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)

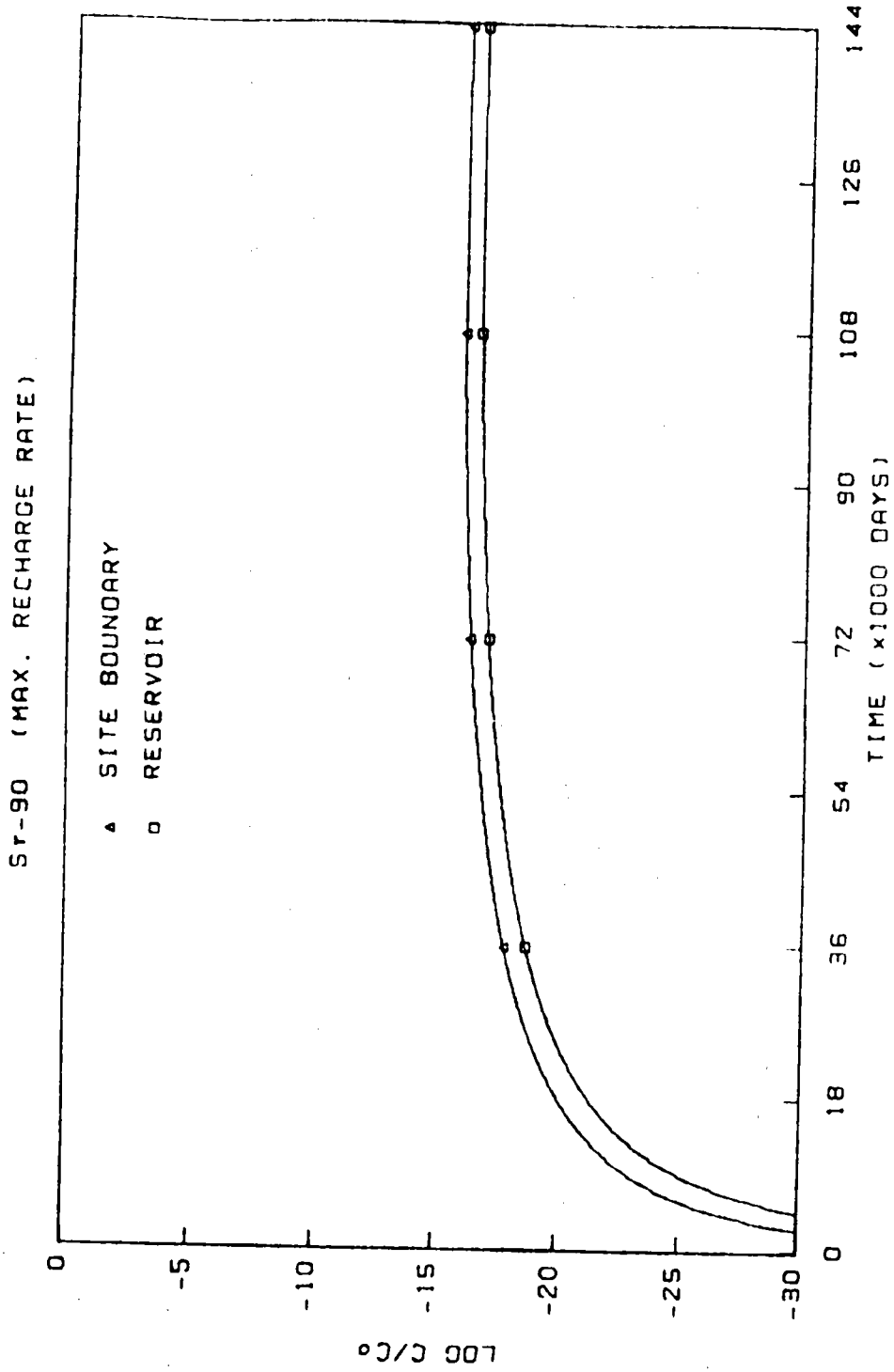

SITE BOUNDARY
RESERVOIR
LOG C/C₀
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x 1000 DAYS)


Sr-90 (B)rock =0.2
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)


Sr-90 (AL=10.AT=1)rock
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (×1000 DAYS)


Sr-90 (AL=100,AT=10)rock
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)


Sr-90 (MIN. RECHARGE RATE)
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)


Sr-90 (MAX. RECHARGE RATE)
SITE BOUNDARY
RESERVOIR
LOG C/Co
0
-5
-10
-15
-20
-25
-30
0
18
36
54
72
90
108
126
144
TIME (x1000 DAYS)